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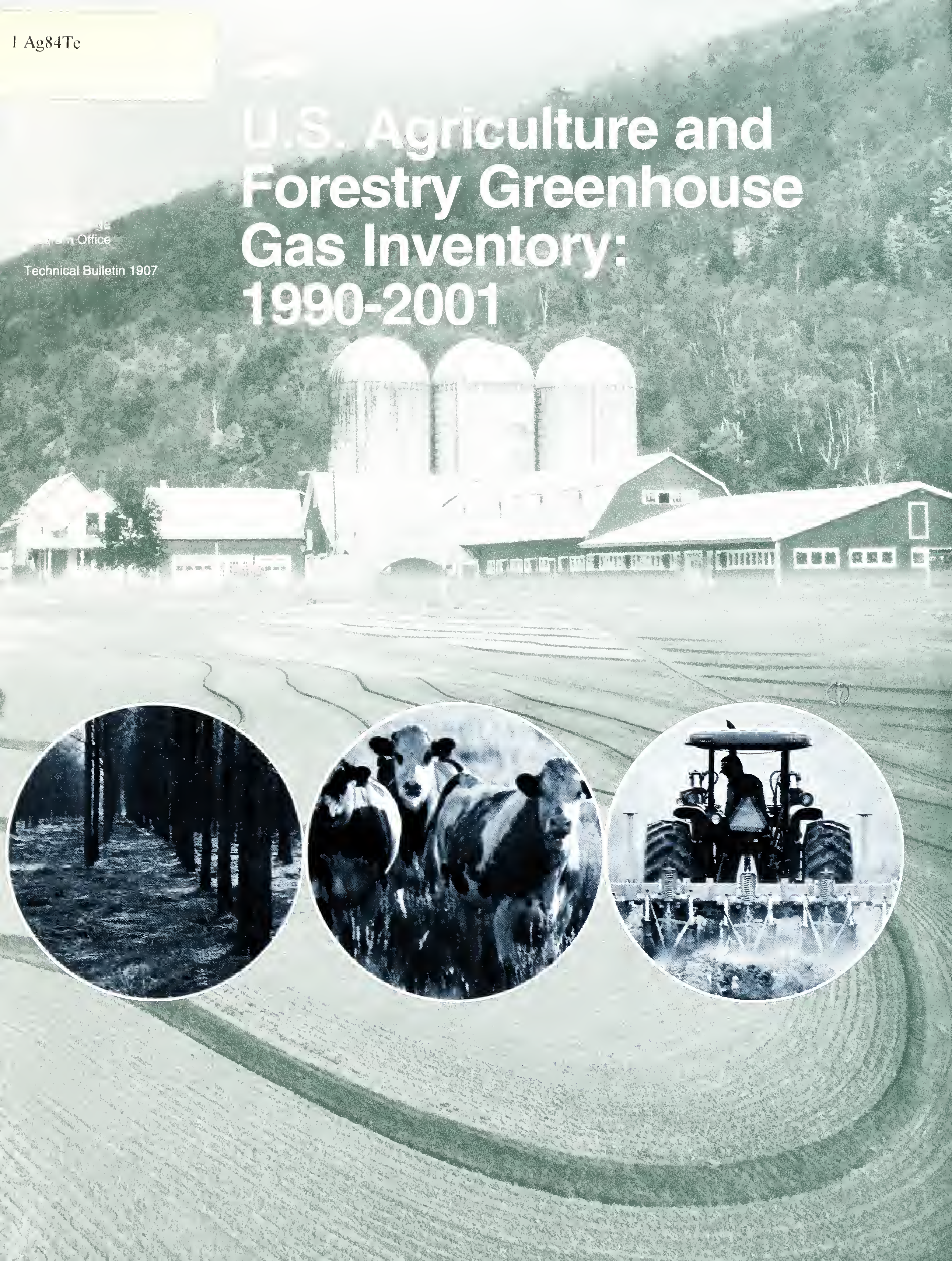
U.S. Agriculture and Forestry Greenhouse Gas Inventory: 1990-2001



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Abstract

The U.S. Agriculture and Forestry Greenhouse Gas Inventory: 1990-2001 (USDA GHG Inventory) is a comprehensive assessment of greenhouse gas emissions and sinks in U.S. agriculture and forests. The USDA GHG Inventory provides extensive, in-depth emissions and sinks estimates for livestock, cropland, and forests, as well as energy consumption in livestock and cropland agriculture. Estimates are provided at State, regional, and national scales, categorized by land ownership and management practices where possible. Information in the report can be used to identify opportunities to reduce emissions and enhance sinks through agriculture and forest management. The report was prepared collaboratively with contributions from the United States Department of Agriculture (Forest Service, Natural Resources Conservation Service, Agricultural Research Service, Office of Energy Policy and New Uses, and the Global Change Program Office), the Environmental Protection Agency (EPA), and researchers at Colorado State University. The estimates in the USDA GHG Inventory are consistent with those published by the EPA in the official Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2001 and submitted to the United Nations Framework Convention on Climate Change in April 2003.

Keywords: greenhouse gases, land use, land use change, carbon stocks, carbon sequestration, enteric fermentation, waste management, soil management, residue burning, rice cultivation, energy consumption.

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Acknowledgments

This report was made possible by contributions from a number of individuals and collaboration between the U.S. Department of Agriculture and the U.S. Environmental Protection Agency. The U.S. Agriculture and Forestry Greenhouse Gas Inventory: 1990-2001 (USDA GHG Inventory) is supplemental to the official Inventory of U.S. Greenhouse Gas Emissions and Sinks (U.S. GHG Inventory) submitted by EPA to the United Nations Framework Convention on Climate Change each April. We would like to acknowledge the contribution of Tom Wirth, Joe Mangino, and Dina Kruger of EPA's Office of Atmospheric Programs, who provided detailed emissions data for the agricultural sources of methane and nitrous oxide reported in Chapters 2 and 3. The methodological discussions for agricultural sources of methane and nitrous oxide draw directly from the official U.S. GHG Inventory. The Natural Resources Ecology Laboratory (NREL) of Colorado State University prepares the agricultural soil carbon estimates for the official U.S. GHG Inventory. NREL also provided the analysis for this report, including State-level estimates of soil carbon stock changes and supporting text.

Several individuals in the U.S. Department of Agriculture contributed to this report. John Kimble and John Brenner from the Natural Resources Conservation Service and Ron Follett from the Agricultural Research Service worked closely with NREL to develop the soil carbon estimates used in this report and to provide additional material on soil carbon stocks in agriculture. Arvin Mosier of the Agricultural Research Service contributed to sections on agricultural sources of methane and nitrous oxide emissions. James Smith, Peter Woodbury, and Linda Heath of the U.S. Forest Service provided Chapter 4, Forest Carbon Sequestration and Products Storage. The U.S. Forest Service provides forest carbon estimates for the official U.S. GHG Inventory. They tailored their analysis for this report to enable a look at regional and land ownership trends in forest carbon. Finally, James Duffield and Hosein Shapouri of the Office of Energy Policy and New Uses prepared Chapter 5, Energy Use in Agriculture. The estimates presented in Chapter 5 are unique to this report and derive from ongoing work in the Office of Energy Policy and New Uses to track fuel consumption in agriculture. Estimates were developed using regional emission factors for electricity consumption provided by Dick Richards from the Department of Energy's Energy Information Administration.

The USDA Global Change Task Force provided support and guidance; comments provided by reviewers, many of whom are on the Global Change Task Force, offered a number of helpful suggestions. Raymond Bridge, Office of the Chief Economist, and the USDA Office of Communications provided assistance with publishing.

Glossary of Terms and Units

CO_2	Carbon dioxide
CH_4	Methane
N_2O	Nitrous oxide
NOx	Nitrogen oxides
C	Carbon
GHG	Greenhouse gas
GWP	Global warming potential
Tg	Teragram (10^{12} grams)
Tg CO_2 eq.	Teragrams of carbon dioxide equivalent units
Gg	Gigagram (10^{9} grams)
Mg	Megagram (10^{6} grams)
t	Metric ton (1,000 kg)
ha	Hectares
ac	Acres
DE	Digestible energy (percent)
Y_m	Fraction of gross energy converted to CH_4
TDN	Total digestible nutrients
VOCs	Volatile organic compounds
VS	Volatile solids
DM	Dry matter
Btu	British thermal unit
Qbtu	Quadrillion British thermal units
Tbtu	Trillion British thermal units
EF	Emission factor
MCF	Methane conversion factor

Chapter 1: Introduction

1.1 Purpose of this Report

The U.S. Agriculture and Forestry Greenhouse Gas Inventory: 1990-2001 was developed to provide a comprehensive assessment of the contribution of U.S. agriculture and forestry to greenhouse gas (GHG) emissions. The document was prepared to support and expand on information provided in the official Inventory of U.S. GHG Emissions and Sinks (U.S. GHG Inventory), which is prepared annually by the U.S. Environmental Protection Agency to meet U.S. commitments under the United Nations Framework Convention on Climate Change (UNFCCC) (EPA 2003a). The U.S. Agriculture and Forestry GHG Inventory: 1990-2001 (USDA GHG Inventory) supplements the U.S. GHG Inventory, providing an in-depth look at agriculture and forestry emissions and sinks of GHGs and presenting additional information on GHG emissions from fuel consumption on U.S. farms.

The U.S. GHG Inventory provides national-level estimates of emissions of the primary GHGs (carbon dioxide, methane, nitrous oxide, and fluorine containing halogen substances) across a broad range of sectors (energy, industrial processes, solvent use, agriculture, land use change and forestry, and waste). Due to the national-level scale of reporting in the U.S. GHG inventory, the report does not provide extensive regional or State GHG emissions data for any one sector. However, in some cases, State and regional emissions data are part of the inventory development process and can be used for more disaggregated analyses. This report customizes the data from the U.S. GHG Inventory in a manner that is useful to agriculture and forestry producers and related industries, natural resource and agricultural professionals, as well as technical assistance providers, researchers, and policymakers. The analyses presented in this report are the result of a collaborative process and direct contributions from EPA, USDA (Forest Service, Natural Resources Conservation Service, Agricultural Research Service, Office of Energy Policy and New Uses, and the Global Change Program Office), and the Natural Resources Ecology Laboratory (NREL) of Colorado State University.

USDA administers a portfolio of conservation programs that have multiple environmental benefits including reductions in GHG emissions and increases in carbon sequestration. In June 2003, Secretary of Agriculture Ann M. Veneman announced a series of measures to incorporate GHG considerations into USDA conservation programs. The information provided in this inventory will be useful in improving our understanding of the magnitude of GHG emissions by State, region, and land use, and by crop, pasture, range, and forest management systems.

This and future USDA GHG Inventory reports will facilitate tracking of progress in promoting carbon sequestration and reducing GHG emissions through agriculture and forest management. This year, the USDA GHG Inventory describes the role of agriculture and forestry in GHG emissions and sinks, including discussions of GHG emissions reductions and carbon sequestration through agriculture and forest management. Extensive and in-depth emissions estimates are presented for all agricultural and forestry GHG sources and sinks for which internationally recognized methods are available. In many cases, emissions estimates are provided at State and regional scales in addition to the national levels provided in the U.S. GHG

Inventory. Emissions are categorized by additional information such as land ownership and management practices where possible. This report will help to:

- Quantify current levels of emissions and sinks at State, regional, and national scales in agriculture and forestry,
- Identify activities that are driving GHG emissions and sinks and trends in these activities, and
- Assess information needed to improve estimates of GHG emissions and sinks.

1.2 Overview of the Report Structure

The report provides detailed trends in agriculture and forestry GHG emissions and sinks, with information by source and sink at State and regional levels. The report is structured mainly from a land use perspective, addressing livestock operations, croplands, and forests separately; but it also includes a chapter on energy use. The livestock chapter inventories GHG emissions from livestock and livestock waste stored and managed in confined livestock operations as well as pasture and range operations. The cropland agriculture chapter addresses emissions from cropland soil amendments, rice production, and residue burning, as well as carbon sequestration in agricultural soils. The forest chapter details carbon sequestration in forest biomass and soils, urban trees, and wood products. Emissions of methane and nitrous oxide in forestry are not addressed since little information is currently available to develop estimates for these sources. The energy chapter provides information on carbon dioxide emissions from energy consumption on U.S. farms, covering GHG emissions from fuel use in livestock and cropland agriculture. While the U.S. GHG Inventory provides estimates of GHG emissions from energy consumption in the production of fertilizer, this indirect source of agricultural GHG emissions is not covered in this report.

Each chapter presents a summary of sources of greenhouse gas emissions and sinks in the land use or category of emissions covered by the chapter. A brief summary of GHG emissions at the national level is provided initially, followed by more detailed descriptions of emissions by each source at national and sub-national scales where available. Detailed accounts of the methods are provided for each source as well as information on the main sources of uncertainty in the methods. In some cases, text describing the methods and uncertainty are taken directly from the U.S. GHG Inventory, with permission from the EPA. In addition, some chapters include discussions of opportunities to mitigate greenhouse gas emissions, additional research underway for refining estimates, and future research priorities for improving estimation techniques.

The remainder of this section provides an overview of greenhouse gas emissions in agriculture and forestry from a global perspective and a summary of greenhouse gas emissions in U.S. agriculture and forestry from 1990 to 2001.

1.3 Global Change and Global Greenhouse Gas Emissions in Agriculture and Forestry

Global concentrations of GHGs in the atmosphere have increased measurably over the past 250 years. Carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) concentrations in the atmosphere have increased by roughly 31 percent, 151 percent, and 17 percent, respectively, since 1750 (IPCC 2001). Agriculture and forestry practices have contributed greenhouse gases to the atmosphere over this time period through fuel consumption, land use conversions, cultivation and fertilization of soils, production of ruminant livestock, and management of livestock manure. The management of agriculture and forest land has helped offset GHG emissions by promoting the biological uptake of CO_2 through the incorporation of carbon into biomass, wood products, and soils.

Observed global increases in CO_2 emissions are primarily a result of fossil fuel combustion at present, but in the past, land use and land use change were major contributors. Currently, land use change is the second largest global cause of CO_2 emissions (IPCC 2001). Land use and land use change can be managed to rebuild carbon stocks in soil and biomass with the potential to essentially reverse past emissions from historical land use conversions. While land use conversion is a large global source of CO_2, within the United States, net forestland area has remained relatively stable for the last century with a relatively small net loss of roughly 4.2 million hectares (Kimble et al. 2003). Since the mid-20thcentury, forestland conversions have been primarily for development purposes. Forest conversions to cropland, pasture, and rangeland were more common prior to 1953 (Kimble et al. 2003).

Over half of global annual emissions of CH_4 and roughly a third of global annual emissions of N_2O are believed to derive from human sources, mainly from agriculture (IPCC 2001). Agricultural activities contribute to these emissions in a number of ways. While losses of N_2O to the atmosphere occur naturally as a result of the soil nitrogen cycle, the application of nitrogen to amend soil fertility can increase the rate of emissions. The rate is amplified when more nitrogen is applied than can be used by the plants. In agricultural practices, nitrogen is added to soils through the use of synthetic fertilizers, application of manure, cultivation of nitrogen-fixing crops/forages (e.g., legumes), and retention of crop residues. Rice cultivation uses periodic flooding of rice paddies, which promotes anaerobic decomposition of organic matter in soil such as rice residue and organic fertilizers by CH_4-emitting soil microbes. Finally, burning of residues in agricultural fields produces CH_4 and N_2O as by-products.

In addition, livestock and livestock waste cause CH_4 and N_2O emissions to the atmosphere. Ruminant livestock such as cattle, sheep, goats, buffalo, and camels emit CH_4 as a byproduct of their digestive processes (called "enteric fermentation"). Livestock waste can release both CH_4 through the biological breakdown of organic compounds and N_2O through nitrification and denitrification of nitrogen contained in manure; the magnitude of emissions depends in large part on manure management practices and to some degree on the energy content of livestock feed.

Agriculture and forest management can offset GHG emissions by increasing capacity for carbon uptake and storage in biomass, wood products, and soils. This process is referred to as carbon sequestration. The net flux of CO_2 between the land and the atmosphere is a balance between carbon losses from land use conversion and land management practices, and carbon gains from forest growth and sequestration in soils (IPCC 2001). Improved forest regeneration and management practices such as density control, nutrient management, and genetic tree

Box 1-1

Conventional Units for Reporting GHG Emissions

The USDA GHG Inventory report follows the international convention for reporting greenhouse gas emissions, as described in the introduction of the U.S. GHG Inventory (EPA 2003). Emissions of greenhouse gases are expressed in equivalent terms, normalized to carbon dioxide using Global Warming Potentials (GWPs) published by the IPCC (IPCC 1995). GWPs, which are based on physical and chemical properties of gases, represent the relative effect of a greenhouse gas on the climate, integrated over a given time period, relative to a single gas (IPCC 2001). The GWP values used in the U.S. GHG Inventory and this report are recommended by the IPCC for national greenhouse gas inventory reporting. These values are referenced to carbon dioxide and based on a 100-year time period (IPCC 1996):

Gas	Atmospheric lifetime (years)	GWP*
CO_2	50-200	1
CH_4	12	21
N_2O	120	310

*For consistency with international reporting standards, the U.S. GHG Inventory uses GWP values published in the IPCC Second Assessment Report (1996). GWP values and estimated atmospheric lifetime were revised for some gases in the IPCC Third Assessment Report (2001).

In the USDA and U.S. GHG Inventories, carbon dioxide equivalent units are expressed in teragrams, where a teragram (10^{12} grams) equals 1 million metric tons. The formula for converting gigagrams (1 gigagram = 10^9 grams) of a greenhouse gas to teragrams of carbon dioxide equivalent (Tg CO_2 eq.) is provided in the U.S. GHG Inventory and is repeated here for clarity:

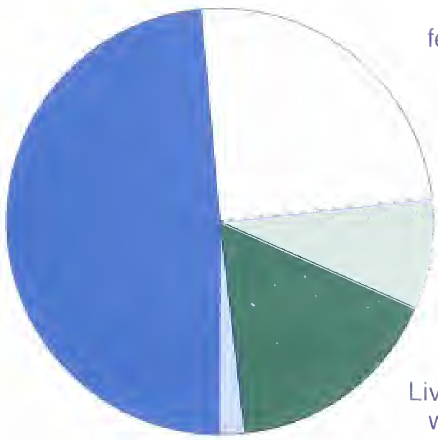
$$\mathrm{TgCO_2\ eq.} = (\mathrm{Gg\ of\ gas}) \times (GWP) \times \left({}^{1Tg}\!\big/_{1{,}000Gg} \right)$$

In the land use sector, where carbon dioxide gas is sequestered and stored as carbon in biomass and soils, greenhouse gas removals are often expressed in units of million metric tons of carbon equivalent (MMTCE). The formula below shows how to convert MMTCE to Tg CO_2 eq. and is based on the molecular weights of carbon and carbon dioxide.

$$\mathrm{TgCO_2\ eq.} = \mathrm{MMTCE} \times \left({}^{44}\!\big/_{12} \right)$$

improvement promote tree growth and result in additional carbon accumulation in biomass. In addition, wood products harvested from forests can serve as long-term carbon storage pools. The adoption of agroforestry practices like windbreaks and riparian forest buffers, which incorporate trees and shrubs into ongoing farm operations, represents a potentially large GHG sink nationally. In addition, agricultural practices such as conservation tillage and grassland practices such as rotational grazing can also reduce carbon losses and promote carbon sequestration in agricultural soils. These practices offset CO_2 emissions caused by land use activities such as conventional tillage and cultivation of organic soils.

Figure 1-1
Agricultural sources of greenhouse gas emissions in 2001



Agriculture and forestry provide opportunities to reduce GHG emissions through targeted management. Innovative practices to reduce GHG emissions from livestock include modifying energy content of livestock feed, inoculating feed with agents that reduce CH_4 emissions from digestive processes, and managing manure in controlled systems that reduce or eliminate GHG emissions. For example, anaerobic digesters are a promising technology for capturing and using CH_4 emissions from livestock waste as an alternative energy source. In addition, GHG emission from soils can be reduced with improved nitrogen use efficiency, involving both reduced nitrogen applications and improved nitrogen uptake by plants. These and other practices, many of which have additional benefits beyond GHG emission reductions, are discussed further in this report.

1.4 Summary of U.S. Greenhouse Gas Emissions in Agriculture and Forestry: 1990-2001

GHG emissions estimates reported here and in the U.S. GHG Inventory are in units of CO_2 equivalents. Box 1-1 describes this reporting convention, which normalizes all GHG emissions to CO_2 equivalents using Global Warming Potentials (GWP). Agriculture in the United States, including livestock, poultry, and crop production, contributed a total of 460 Tg CO_2 eq. (teragrams of carbon dioxide equivalents) to the atmosphere in 2001 (Table 1-1). This total includes an offset from agricultural soil carbon sequestration of roughly 15 Tg CO_2 eq. Agriculture accounted for close to 7 percent of U.S. GHG emissions across all sectors (total U.S. emissions, 6,936 Tg CO_2 eq.) (EPA 2003a). Forestry in the United States, including carbon sequestration in forest trees and soils, wood products, and urban trees, contributed a net

Table 1-1 Summary of agriculture and forestry GHG emissions and sinks, 1990, 1995-2001

	1990	1995	1996	1997	1998	1999	2000	2001
				Tg CO_2 eq.				
Livestock	225.03	240.09	236.61	234.44	234.12	233.28	231.38	230.73
Enteric fermentation, CH_4	117.90	123.00	120.50	118.30	116.70	116.60	115.70	114.80
Waste, CH_4	31.34	36.25	34.95	36.63	39.10	38.98	38.35	39.01
Waste, N_2O	75.79	80.85	81.16	79.52	78.32	77.70	77.33	76.92
Crops	202.83	213.57	223.50	230.73	235.86	234.25	230.14	229.10
Cropland soils, N_2O	207.93	219.77	229.00	235.93	238.16	236.65	235.14	235.40
Rice cultivation, CH_4	7.10	7.60	7.00	7.50	7.90	8.30	7.50	7.60
Residue burning, CH_4	0.70	0.70	0.70	0.80	0.80	0.80	0.80	0.80
Residue burning, N_2O	0.40	0.40	0.40	0.40	0.50	0.40	0.50	0.50
Agricultural soils[1] CO_2	(13.30)	(14.90)	(13.60)	(13.90)	(11.50)	(11.90)	(13.80)	(15.20)
Forestry	(1041.40)	(1037.70)	(1037.70)	(817.70)	(810.30)	(821.30)	(814.00)	(817.70)
Forests, CO_2	(773.70)	(773.70)	(773.70)	(546.30)	(546.30)	(546.30)	(546.30)	(546.30)
Wood products, CO_2	(209.00)	(205.30)	(205.30)	(212.70)	(205.30)	(216.30)	(209.00)	(212.70)
Urban trees, CO_2	(58.70)	(58.70)	(58.70)	(58.70)	(58.70)	(58.70)	(58.70)	(58.70)
Energy use, CO_2	NA	NA	NA	NA	NA	NA	NA	111.09
Net emissions from agriculture and forestry[2]	(613.55)	(584.04)	(577.59)	(352.53)	(340.32)	(353.76)	(352.48)	(357.87)
Agriculture	427.85	453.66	460.11	465.17	469.98	467.54	461.52	459.83
Forestry	(1041.40)	(1037.70)	(1037.70)	(817.70)	(810.30)	(821.30)	(814.00)	(817.70)

Note: Parentheses indicate net sequestration.
[1] Agricultural soil carbon sequestration includes sequestration on land set aside under the Conservation Reserve Program (CRP) and on range/grazing lands, in addition to cultivated mineral and organic soils.
[2] Does not include emissions from energy use.
NA - Not Available.

reduction from the atmosphere of 818 Tg CO_2 eq. in 2001, which is a near 12-percent offset in total U.S. GHG emissions (EPA 2003a).[1] Agriculture and forestry carbon sequestration more

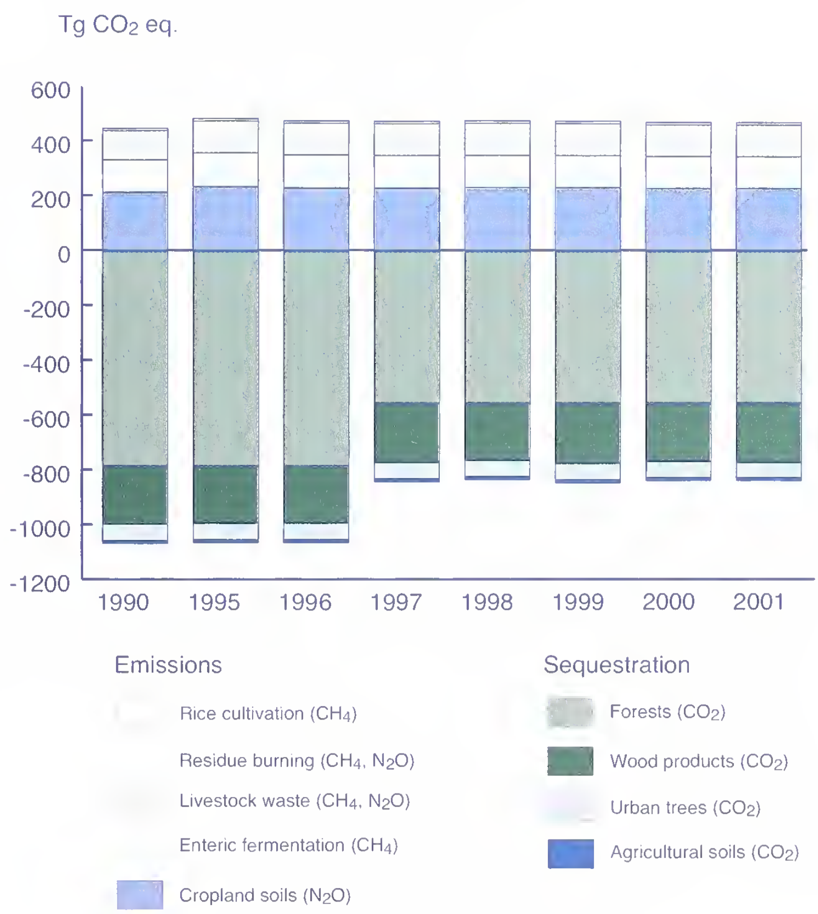
[1] This total does not include landfilled yard trimmings, which is included in the U.S. GHG Inventory, because this sink source is not managed or influenced by forest and agricultural practices. Landfilled yard trimmings contribute a small portion to overall sequestration of 5 Tg CO_2 eq.

than offset the total GHG emissions, with a net sequestration of 358 Tg CO_2 eq.

One-half of agriculture's GHG emissions in 2001 were due to crop production. Forty-eight percent were from nutrient amendments to cropland soils (Figure 1-1). Other crop sources—rice cultivation and residue burning—contributed 2 percent of overall agricultural emissions. Livestock production is responsible for the remaining half of agricultural emissions, with nearly 25 percent from enteric fermentation and 25 percent from waste.

Overall emissions profiles of the predominant agriculture and forestry sources and sinks appear relatively constant since 1990 with a few notable trends (Figure 1-2). Annual agricultural emissions from both livestock and crop production increased by 34 Tg CO_2 eq., or about 8 percent, since 1990 (Figure 1-2), with the largest increases from crop production. N_2O from nutrient amendments to cropland soils increased to a peak in 1998 and has declined slightly since; but by 2001, emissions from this source were 27 Tg CO_2 eq. greater than 1990 levels (Table 1-1). The second greatest increase in emissions was seen in livestock production. Emissions from livestock waste were 8 percent greater in 2001 than in 1990 (Table 1-1). Emissions from enteric fermentation have declined slightly over the same time period. Net sequestration, which includes carbon sequestration in agricultural soils and forests, declined by about 20 percent between 1990 and 2001, resulting primarily from a decline in the rate of net carbon accumulation in forests (EPA 2003a). An

Figure 1-2
Agriculture and forestry GHG emissions and sequestration for 1990, 1995-2001



Note: Negative values correspond to sequestration.

apparent abrupt change in net sequestration reported from 1996 to 1997 is a result of underlying data sources; the decline was likely more gradual through time. In contrast, annual carbon accumulation in agricultural soils increased from 1990 to 2001 by 2 Tg CO_2 eq.; wood products carbon storage pools also increased during this time (Table 1-1).

Annual CO_2 emissions from energy use in agriculture are small relative to total energy use across all sectors in the United States. In 2001, fuel and electricity consumption associated with crop and livestock operations resulted in 111 Tg CO_2 (Table 1-1), which is about 2 percent of overall energy-related emissions for 2001 (5,597 Tg CO_2). Electricity use led to about 53 percent of CO_2 emissions from energy use in agriculture; diesel fuel use led to about 30 percent, while gasoline, natural gas, and liquefied petroleum gas contributed less than 10 percent each to total CO_2 emissions from energy use in agriculture.

Chapter 2: Livestock Emissions

2.1 Sources of Greenhouse Gas Emissions in Livestock

Livestock contribute greenhouse gas emissions (GHG) to the atmosphere both directly and indirectly. Livestock emit methane (CH_4) directly as a byproduct of digestion through a process called enteric fermentation. In addition, livestock manure and urine ("waste") cause CH_4 and nitrous oxide (N_2O) emissions to the atmosphere as a result of decomposition and nitrification/ denitrification. This chapter provides national and State-level data on CH_4 emissions from enteric fermentation, and on CH_4 and N_2O emissions from livestock waste. State-level livestock population data also are presented in this chapter because of the relationships between GHG emissions from livestock and livestock population sizes.

2.1.1 Enteric Fermentation

Enteric fermentation is a normal digestive process where microbial populations in the digestive tract break down food and cause animals to excrete CH_4 gas as a by-product. CH_4 is then emitted from the animal to the atmosphere thorough exhaling or eructation. Ruminant livestock, including cattle, sheep, and goats, have greater rates of enteric fermentation because of their unique digestive system, which includes a large rumen or fore-stomach where enteric fermentation takes place. Non-ruminant livestock such as swine, horses, and mules produce less CH_4 from enteric fermentation because it takes place in the large intestine, which has a smaller capacity than the rumen. The energy content and quantity of animal feed also affect the amount of CH_4 produced in enteric fermentation, with lower quality and higher quantities of feed causing greater CH_4 emissions.

2.1.2 Livestock Waste

Livestock waste is "unmanaged" when it is deposited directly on pastures, range, or paddock. Alternatively, livestock waste can be "managed" in storage and treatment systems, or spread daily on fields in lieu of long-term storage. Many livestock producers in the U.S. manage livestock waste in systems such as solid storage, dry lots, liquid-slurry storage, deep pit storage, and anaerobic lagoons. Table 2-1 provides descriptions of managed and unmanaged pathways for livestock waste, indicating in general terms the impacts of different pathways on GHG emissions. Sometimes livestock waste that is stored and treated is subsequently applied as a nutrient amendment to agricultural soils. GHG emissions from the application of treated waste to soils as a nutrient amendment are discussed in the next chapter along with GHG emissions from other nutrient amendments for crop production.

The magnitude of CH_4 and N_2O emissions from managed livestock waste depends on environmental conditions. CH_4 is emitted under conditions that promote anaerobic decomposition, occurring when oxygen is not available to bacteria responsible for waste breakdown, forcing an alternate metabolic pathway that creates CH_4 as a by-product. Storage in ponds, tanks, or pits such as those that are coupled with liquid/slurry flushing systems often promote anaerobic conditions (i.e., where oxygen is not available and CH_4 is produced) whereas solid waste stored in stacks or pits tends to provide aerobic conditions (i.e., where

Table 2-1 Description of livestock waste deposition and storage pathways

Management	Description	Relative CH_4 emissions	Relative N_2O emissions
Pasture/range/paddock	Waste from pasture and range grazing animals is deposited directly onto the soil.	low	high
Daily spread	Waste is collected and spread on fields. There is little or no storage of the waste before it is applied to soils.	low	zero
Solid storage	Waste (with or without litter) is collected by some means and placed under long-term bulk storage.	low	high
Dry lot	Waste is deposited directly onto unpaved feedlots where the manure is allowed to dry and is periodically removed (after removal it is sometime spread onto fields).	low	high
Liquid/slurry	Waste is collected and transported in a liquid state to tanks for storage. The liquid/slurry mixture may be stored for a long time and water may be added to facilitate handling.	moderate to high	low
Anaerobic lagoon	Waste is collected using a flush system and transported to lagoons for storage. Waste resides in lagoons for 30-200 days.	variable	low
Pit storage	Waste is stored in pits below livestock confinements.	moderate to high	low
Poultry house with bedding	Waste is excreted on poultry house floor covered with bedding; poultry can walk on the floor.	low	high
Poultry house without bedding	Waste is excreted on poultry house floor, which is not covered with bedding; poultry cannot walk on the floor.	low	low

Source: adapted from IPCC 2000.

oxygen is available and CH_4 is not produced). High temperatures generally accelerate the rate of decomposition of organic compounds in waste, increasing CH_4 emissions under anaerobic conditions. In addition, longer residency time in a storage system can increase CH_4 production, and moisture additions, particularly in solid storage systems that normally experience aerobic conditions, can amplify CH_4 emissions.

While environmental conditions of waste storage and handling are important factors affecting CH_4 emissions, diet and feed characteristics are also important determinants. Livestock feed, diet, and growth rates affect both the amount and quality of manure produced per animal. Not only do greater amounts of manure lead to more CH_4 being emitted, but higher energy feed also produces manure with more volatile solids, increasing the substrate from which CH_4 is produced. However, this impact is somewhat offset by the possibility of achieving higher digestibility in feeds, and thus less waste energy.

Table 2-2 U.S. GHG emissions by livestock and source in 2001

	Enteric fermentation CH_4	Livestock waste CH_4	Livestock waste N_2O[1]	Livestock waste (indirect) N_2O[2]
		Tg CO_2 eq.		
Dairy cattle	26.90	15.20	5.09	--
Beef cattle	82.70	3.26	41.81	--
Swine	1.90	17.17	0.62	--
Poultry	0.20	2.70	7.41	--
Goats		0.01	0.20	--
Horses	2.00	0.63	2.54	--
Sheep	1.20	0.04	0.28	--
Total	114.90	39.01	57.95	18.97

[1] N_2O from managed livestock waste and unmanaged waste, direct emissions only.
[2] N_2O from leaching/run-off and volatilization of unmanaged manure deposited on pasture, range, and paddock. Estimates are not available by livestock category.

The production of N_2O from managed livestock waste depends on the composition of the waste, the type of bacteria involved, and the conditions following excretion. For N_2O emissions to occur, the waste must first be handled aerobically where ammonia or organic nitrogen is converted to nitrates and nitrites (nitrification), and then handled anaerobically where the nitrates and nitrites are reduced to nitrogen gas (N_2), with intermediate production of N_2O and nitric oxide (NO) (denitrification) (Groffman et al. 2000). These emissions are most likely to occur in dry waste handling systems that have aerobic conditions, but that also contain pockets of anaerobic conditions due to saturation. For example, waste in dry lots is deposited on soil, oxidized to nitrite and nitrate, and has the potential to encounter saturated conditions.

Unmanaged livestock waste deposited on pasture, range, or paddock creates N_2O emissions as a result of adding nitrogen to soils. When added to soils, nitrogen provides the initial substrate for the natural cycle of nitrification and denitrification. N_2O is a by-product of this cycle; thus more nitrogen added to soils yields more N_2O released to the atmosphere. Nitrogen is added to soils through deposition of livestock waste directly onto soils. A portion of the deposited nitrogen volatilizes to the atmosphere in various gaseous forms and is eventually re-deposited onto the soils. In addition, some nitrogen in livestock waste leaches into groundwater and surface runoff, creating additional N_2O emissions.

2.2 U.S. Livestock Populations

GHG emissions from livestock are inherently tied to livestock population sizes because the livestock are either directly or indirectly the source for the emissions. Livestock population data are collected annually by USDA's National Agricultural Statistics Service (USDA NASS). Those data are an input into the GHG estimates from livestock in the official U.S. GHG Inventory.

Beef and dairy cattle, swine, sheep, goats, poultry, and horses are raised throughout the United States. Detailed livestock population numbers for each State in 2001 are provided in Appendix Table A-1. Appendix Table A-2 shows total national livestock population sizes from 1990 to 2001 by livestock categories. Trends for beef cattle, dairy cattle, and swine are described in more detail below because of their relatively high population numbers and consequently high contributions to GHG emissions. Poultry populations are also described below because of their proportionally large contribution to N_2O emissions through their waste, although overall emissions from poultry are relatively low.

Texas raised by far the most beef cattle at just over 14 million head in 2001 (Appendix Table A-1). Kansas, Nebraska, Oklahoma, and Missouri each raised over 4 million head of beef cattle, while several other States raised around 2 million head of beef cattle. Fewer dairy cattle than beef cattle were raised in the United States in 2001. Dairy cattle populations were highest in California and Wisconsin, with each State having populations near 2 million (Appendix Table A-1). Minnesota, New York, and Pennsylvania had the next largest populations of dairy cattle, ranging from 750,000 to 950,000 head in each State. Most States had far fewer than 500,000 head of dairy cattle.

Iowa was the largest swine producer with nearly 15 million head in 2001 (Appendix Table A-1). North Carolina housed the second largest swine population at just fewer than 10 million head. Illinois, Indiana, Minnesota, Missouri, Nebraska, and Oklahoma also had sizeable swine populations.

Arkansas and Georgia had the largest poultry populations in 2001, with roughly 250 million head of poultry in each State (Appendix Table A-1). Alabama, North Carolina, Mississippi, and Texas also had large populations of poultry, between 125 and 200 million head each. Michigan, Washington, Maine, New York, and Illinois had poultry populations between 50 and 100 million head.

2.3 Summary of U.S. Greenhouse Gas Emissions from Livestock

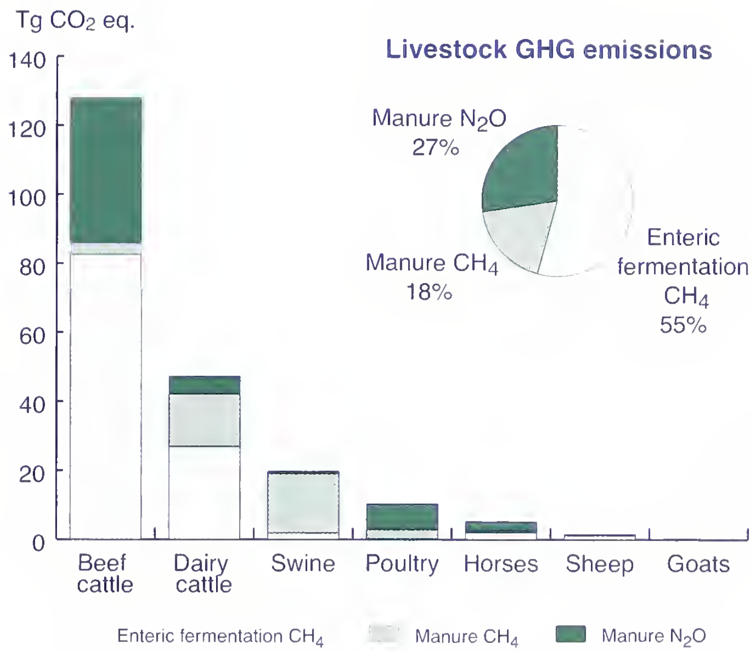
A total of 231 Tg CO_2 eq. of GHG was emitted from livestock and livestock waste in 2001 (Table 2-2). Enteric fermentation and livestock waste sources were nearly equally responsible for these emission, with 115 Tg CO_2 eq. from enteric fermentation and 116 Tg CO_2 eq. from all livestock waste sources combined. Of the emissions from livestock waste, 34 percent were CH_4 (39 Tg CO_2 eq.) and 66 percent were N_2O (77 Tg CO_2 eq.).

Excluding indirect emissions of N_2O from unmanaged livestock waste,[2] beef cattle were responsible for the largest fraction of GHG emissions from livestock in 2001, with the majority

[2] Estimates for this source are not available by livestock category.

of emissions in CH_4 from enteric fermentation (Figure 2-1, Table 2-2). Dairy cattle were the second largest livestock source of GHG emissions, also primarily CH_4 from enteric fermentation. The third largest livestock source was swine, nearly all of which was CH_4 from waste. Poultry, while the fourth largest overall source of livestock emissions, is the second largest source of N_2O emissions, next to beef cattle. Horses, goats, and sheep caused relatively small GHG emissions when compared to other animal groups.

Figure 2-1
U.S. Greenhouse gas emissions from livestock, 2001



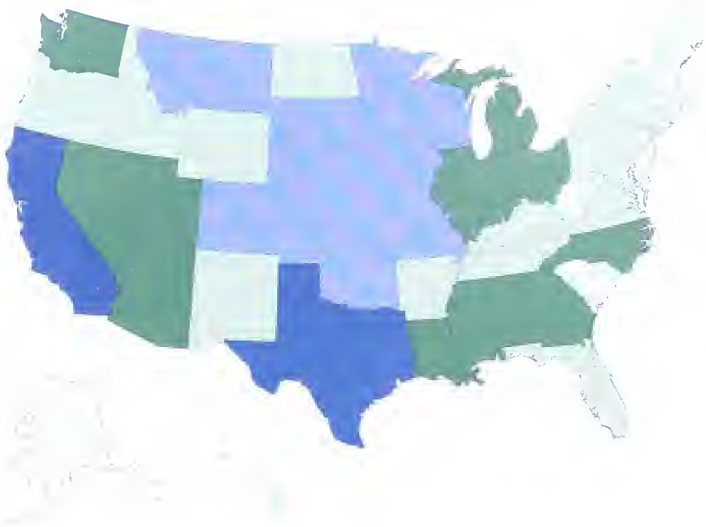
2.4 Enteric Fermentation

Texas and California had the largest aggregate CH_4 emissions from enteric fermentation across all livestock types in 2001 (Map 2-1). Enteric fermentation in Texas released 14 Tg CO_2 eq. of CH_4 in 2001, while in California it led to 7 Tg CO_2 eq. of CH_4 (Appendix Table A-3). These emissions were largely tied to the sizable populations of cattle in both States. However, enteric fermentation emissions in Texas were mostly from beef cattle, whereas in California they were mostly from dairy cattle (Appendix Table A-4). Central, Northern Plains, and some Western States also had relatively high CH_4 emissions from enteric fermentation, ranging between 3 and 6 Tg CO_2 eq. per State in 2001. The smallest emissions of CH_4 from enteric fermentation were found in the Northeast, Hawaii, and Alaska.

Annual emissions of CH_4 from enteric fermentation fluctuated up and down by less than a few Tg CO_2 eq. between 1990 and 2001 (Appendix Table A-3). A continuing trend of decreasing emissions began after 1995, when estimates dropped by about 3 Tg CO_2 eq. (~2 percent of total). Emissions continued to decline, but at a slower rate each year. Overall, by 2001, CH_4 emissions from enteric fermentation declined by about 2.5 percent compared to 1990 levels. State-level annual estimates of methane emissions from enteric fermentation from 1990 to 2001 are provided in Appendix Table A-3. A complete time series of enteric fermentation emissions from all livestock types is shown in Table 2-3.

Map 2-1

Methane emissions from enteric fermentation in 2001



Emission level Tg CO_2 eq.	Percentile*
0 - 0.05	0 - 5
0.06 - 0.60	5 - 25
0.61 - 1.80	25 - 50
1.81 - 2.90	50 - 75
2.91 - 6.40	75 - 95
6.41 - 14.40	95 - 100

* Percentile of the range of emission levels across all states

2.5 Methods for Estimating CH_4 Emissions from Enteric Fermentation

EPA provided USDA with State and national estimates of GHG emissions from enteric fermentation. The estimates were prepared following a method developed by EPA (EPA 1993b), the current version of which is described in Annex L of the U.S. GHG Inventory. USDA data on diet characteristics of livestock populations were used as an input to the estimates, along with emission factors and other parameters developed by EPA, USDA, and others. These data were used in the official U.S. GHG Inventory covering years 1990-2001 (EPA 2003a).

The official U.S. GHG Inventory estimates for enteric fermentation are consistent with the methodological framework provided by the Intergovernmental Panel on Climate Change (IPCC) for preparing national GHG inventories. The IPCC guidance is organized into a hierarchical, tiered structure, where higher tiers correspond to more complex and detailed methodologies. The methods detailed below correspond to both tier 1 and tier 2 approaches. With the permission of EPA, Annex L is recreated below.

2.5.1 Annex L

CH_4 emissions from enteric fermentation were estimated for five livestock categories: cattle, horses, sheep, swine, and goats. Emissions from cattle represent the majority of U.S. emissions; consequently, the more detailed IPCC Tier 2 methodology was used to estimate emissions from cattle and the IPCC Tier 1 methodology was used to estimate emissions from the other types of livestock.

2.5.2 Estimate Methane Emissions from Cattle

This section describes the process used to estimate CH_4 emissions from cattle enteric

Table 2-3 U.S. methane emissions from enteric fermentation, 1990-2001

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
						Tg CO_2 eq.						
Total	117.9	117.1	119.4	118.8	120.4	123	120.5	118.3	116.7	116.6	115.7	114.8
Beef cattle	83.2	82.3	84.7	85.5	87.1	89.7	88.8	86.6	85	84.7	83.5	82.7
Dairy cattle	28.9	28.9	28.9	27.6	27.6	27.7	26.3	26.4	26.3	26.6	27	26.9
Horses	1.9	1.9	1.9	1.9	1.9	1.9	1.9	2	2	2	2	2
Sheep	1.9	1.9	1.8	1.7	1.7	1.5	1.4	1.3	1.3	1.2	1.2	1.2
Swine	1.7	1.8	1.8	1.8	1.9	1.9	1.8	1.8	2	1.9	1.9	1.9
Goats	0.3	0.3	0.3	0.3	0.2	0.2	0.2	0.2	0.2	0.2	0.2	0.2
						Gg						
Total	5,612	5,576	5,685	5,658	5,733	5,855	5,737	5,635	5,557	5,551	5,509	5,468
Beef cattle	3,961	3,920	4,031	4,070	4,147	4,272	4,227	4,124	4,046	4,035	3,976	3,936
Dairy cattle	1,375	1,378	1,375	1,316	1,314	1,320	1,254	1,255	1,251	1,266	1,284	1,282
Horses	91	92	92	92	92	92	93	93	94	93	94	95
Sheep	91	89	86	82	79	72	68	64	63	58	56	56
Swine	81	85	88	87	90	88	84	88	93	90	88	88
Goats	13	13	13	12	12	11	10	10	10	10	10	10

fermentation. A model based on recommendations provided in IPCC/UNEP/OECD/IEA (1997) and IPCC (2000) was developed that uses information on population, energy requirements, digestible energy, and CH_4 conversion rates to estimate CH_4 emissions. The emission methodology consists of the following three steps: (1) characterize the cattle population to account for animal population categories with different emissions profiles; (2) characterize cattle diets to generate information needed to estimate emissions factors; and (3) estimate emissions using these data and the IPCC Tier 2 equations.

Step 1: Characterize U.S. Cattle Population

Each stage in the cattle lifecycle was modeled to simulate the cattle population from birth to slaughter. This level of detail accounts for the variability in CH_4 emissions associated with each life stage. Given that the time in which cattle can be in a stage can be less than 1 year (e.g., beef calves are weaned at 7 months), the stages are modeled on a per-month basis. The type of cattle use also impacts CH_4 emissions (e.g., beef versus dairy). Consequently, cattle life stages were modeled for several categories of dairy and beef cattle. These categories are listed in Appendix Table A-5.

The key variables tracked for each of these cattle population categories except bulls[3] are as follows:

Calving Rates: The number of animals born on a monthly basis was used to initiate monthly cohorts and to determine population age structure. The number of calves born each month was obtained by multiplying annual births by the percentage of births by month. Annual birth information for each year was taken from USDA NASS (Cattle: 2002, 2001, 2000, 1999, and 1995). Average percentages of births by month for beef from USDA APHIS (NAHMS 1998, 1994, 1993) were used for 1990 through 2001. For dairy animals, birth rates were assumed constant throughout the year. Whether calves were born to dairy or beef cows was estimated using the dairy cow calving rate and the total dairy cow population to determine the percent of births attributable to dairy cows, with the remainder assumed to be attributable to beef cows.

Average Weights and Weight Gains: Average weights were tracked for each monthly age group using starting weight and monthly weight gain estimates. Weight gain (i.e., pounds per month) was estimated based on weight gain needed to reach a set target weight, divided by the number of months remaining before target weight was achieved. Birth weight was assumed to be 88 pounds for both beef and dairy animals. Weaning weights were estimated to range from 480 to 575 pounds, depending on birth month. Other reported target weights were available for 12-, 15-, 24-, and 36-month-old animals. Live slaughter weights were derived from dressed slaughter weight data for each year from USDA NASS (Livestock Slaughter: 2002, 2001, 2000; Cattle: 1999 1995). Live slaughter weight was estimated as dressed weight divided by 0.63.

Feedlot Placements: Feedlot placement statistics were available that specify placement of animals from the stocker population into feedlots on a monthly basis by weight class. The model used these data to shift a sufficient number of animals from the stocker cohorts into the feedlot populations to match the reported placement data. After animals are placed in feedlots they progress through two steps. First, animals spend time on a step-up diet to become acclimated to the new feed type. Animals are then switched to a finishing diet for a period of time before they are slaughtered. The length of time an animal spends in a feedlot depends on the start weight (i.e., placement weight), the rate of weight gain during the start-up and finishing phase of diet, and the end weight (as determined by weights at slaughter). Weights vary by cohort. Weight gain during start-up diets is estimated to be 2.8 to 3 pounds per day. Weight gain during finishing diets is estimated to be 3 to 3.3 pounds per day (Johnson 1999). All animals are estimated to spend 25 days in the step-up diet phase (Johnson 1999). Length of time to finishing was calculated based on start weight, weight gain per day, and target slaughter weight. Once animals in the model are placed in the feedlot, they are slaughtered only after they reach the target weight.

[3] Only end-of-year census population statistics and a national emission factor are used to estimate methane emissions from the bull population.

Pregnancy and Lactation: Energy requirements and hence, composition of diets, level of intake, and emissions for particular animals, are greatly influenced by whether the animal is pregnant or lactating. Information is therefore needed on the percentage of all mature animals that are pregnant each month, as well as milk production, to estimate CH_4 emissions. A weighted average percent of pregnant cows each month was estimated using information on births by month and average pregnancy term (model uses a 9-month pregnancy term). For beef cattle, a weighted average total milk production per animal per month was estimated using information on typical lactation cycles and amounts (NRC 1999), and data on births by month. This process results in a range of weighted monthly lactation estimates expressed as lbs/animal/month. The monthly estimates from January to December are 3.33, 5.06, 8.70, 12.01, 13.58, 13.32, 11.67, 9.34, 6.88, 4.45, 3.04, and 2.77 lbs milk/animal/month. Monthly estimates for dairy cattle were taken from USDA monthly milk production statistics.

Death Rates: This factor is applied to all heifer and steer cohorts to account for death loss within the model on a monthly basis. The death rates are estimated by determining the death rate that results in model estimates of the end-of-year population for cows that match the published end-of-year population census statistics. Death rates are assumed to be 0.35 percent for calves (which are only calves for a fraction of the year), 1 percent annually for stockers, and 2 percent annually for replacements. Death rate statistics for beef and dairy cows are calculated each year, based on starting and ending populations and available replacements.

Number of Animals per Category Each Month: The population of animals per category is calculated based on number of births (or graduates) into the monthly age group minus those animals that die or are slaughtered and those that graduate to the next category (including feedlot placements). These monthly age groups are tracked in the enteric fermentation model to estimate emissions by animal type on a regional basis. Regions are defined in Appendix Table A-22.

Animal Characteristic Data: Dairy lactation estimates for 1990 through 2001 are shown in Appendix Table A-6. Appendix Table A-7 provides the target weights used to track average weights of cattle by animal type. Appendix Table A-8 provides a summary of the reported feedlot placement statistics for 2001. Data on feedlot placements were available for 1996 through 2001. Data for 1990 to 1995 were based on the average of monthly placements from the 1996-98 reported figures.

Cattle population data were taken from USDA NASS. Populations upon which all livestock-related emissions are based are in Appendix Table A-2. The USDA NASS publishes monthly, annual, and multi-year livestock population and production estimates. Multi-year reports include revisions to earlier published data. Cattle and calf populations, feedlot placement statistics (e.g., number of animals placed in feedlots by weight class), slaughter numbers, and lactation data were obtained from the USDA NASS (Cattle: 2002 2001, 2000, 1999, 1995;

Livestock Slaughter: 2002, 2001, 2000). Beef calf birth percentages were obtained from the USDA APHIS National Animal Health Monitoring System (NAHMS: 1998, 1994, 1993).

Step 2: Characterize U.S. Cattle Population Diets

To support development of digestible energy (DE, the percent of gross energy intake digestible to the animal) and CH_4 conversion rate (Y_m, the fraction of gross energy converted to CH_4) values for each of the cattle population categories, data were collected on diets considered representative of different regions. For both grazing animals and animals being fed mixed rations, representative regional diets were estimated using information collected from State livestock specialists and from USDA APHIS NAHMS (1996). The data for each of the diets (e.g., proportions of different feed constituents, such as hay or grains) were used to determine chemical composition for use in estimating DE and Y_m for each animal type. Additional detail on the regional diet characterization is provided in EPA (2000a).

DE and Y_m vary by diet and animal type. The IPCC recommends Y_m values of 3.5 to 4.5 percent for feedlot cattle and 5.5 to 6.5 percent for all other cattle. Given the availability of detailed diet information for different regions and animal types in the United States, DE and Y_m values unique to the United States[4] were developed.

Appendix Table A-9 shows the regional DE, the Y_m, and percent of total U.S. cattle population in each region based on 2001 data. DE and Y_m values were estimated for each cattle population category, for each year in the time series based on physiological modeling, published values, and/or expert opinion.

DE and Y_m values for dairy cows were estimated using a model (Donovan and Baldwin 1999) that represents physiological processes in the ruminant animals. The three major categories of input required by the model are animal description (e.g., cattle type, mature weight), animal performance (e.g., initial and final weight, age at start of period), and feed characteristics (e.g., chemical composition, habitat, grain or forage). Data used to simulate ruminant digestion is provided for a particular animal that is then used to represent a group of animals with similar characteristics. The model accounts for differing diets (i.e., grain-based, forage-based, range-based), so that Y_m values for the variable feeding characteristics within the U.S. cattle population can be estimated.

To calculate the DE values for grazing beef cattle, the diet descriptions were used to estimate weighted DE values for a combination of forage only and supplemented diets. Where DE values were not available for specific feed types, total digestible nutrients (TDN) as a percent of dry matter (DM) intake was used as a proxy for DE as it is essentially the same as the DE value.

[4] In some cases, the Y_m values used for this analysis extend beyond the range provided by the IPCC. However, EPA believes that these values are representative for the United States due to the research conducted to characterize the diets of U.S. cattle and to assess the Y_m values associated with different animal performance and feed characteristics in the United States.

For forage diets, two separate regional DE values were used to account for the generally lower forage quality in the western United States. For non-western grazing animals, the forage DE was an average of the seasonal "TDN percent DM" for Grass Pasture diets listed in Appendix Table 1 of the NRC (2000). This average DE for the non-western grazing animals was 64.7 percent. This value was used for all regions except the West. For western grazing animals, the forage DE was calculated as the average "TDN percent DM" for meadow and range diets listed in Appendix Table 1 of the NRC (2000). The calculated DE for western grazing animals was 58.5 percent. The supplemental diet DE values were estimated for each specific feed component, as shown in Appendix Table A-10, along with the percent of each feed type in each region. Finally, weighted averages were developed for DE values for each region using both the supplemental diet and the forage diet.[5] For beef cows, the DE value was adjusted downward by 2 percent to reflect the reduced diet of the mature beef cow. The percent of each diet that is assumed to be supplemental and the DE values for each region are shown in Appendix Table A-11. Y_m values for all grazing beef cattle were set at 6.5 percent based on Johnson (2002).

For feedlot animals, DE and Y_m values for 1996 through 2001 were taken from Johnson (1999). Values for 1990 through 1995 were linearly extrapolated from the 1996 value based on Johnson (1999). Feedlot and dairy cow DE are assumed to be slightly less efficient prior to1996 to reflect changes in feed quality; as a result Y_m is also assumed to be higher in those earlier years. In response to peer reviewer comments (Johnson 2000), values for dairy replacement heifers are based on EPA (1993b).

Step 3: Estimate Methane Emissions from Cattle

Emissions were estimated in three steps: a) determine gross energy intake using the IPCC (2000) equations, b) determine an emissions factor using the GE values and other factors, and c) sum the daily emissions for each animal type. The necessary data values include:

- Body weight (kg)
- Weight gain (kg/day)
- Net energy for activity (C_a)[6]
- Standard reference weight (dairy = 1,324 lbs; beef = 1,195 lbs)[7]
- Milk production (kg/day)
- Milk fat (percent of fat in milk = 4)
- Pregnancy (percent of population that is pregnant)
- DE (percent of gross energy intake digestible)
- Y_m (the fraction of gross energy converted to CH_4)

[5] For example, in California the forage DE of 64.7 was used for 95 percent of the grazing cattle diet and a supplemental diet DE of 65.2 percent was used for 5 percent of the diet, for a total weighted DE of 64.9 percent.
[6] Zero for feedlot conditions, 0.17 for high-quality confined pasture conditions, 0.36 for extensive open range or hilly terrain grazing conditions. C_a factor for dairy cows is weighted to account for the fraction of the population in the region that grazes during the year.
[7] Standard reference weight is used in the model to account for breed potential.

Step 3a: Gross Energy, GE

As shown in the following equation, Gross Energy (GE) is derived based on the net energy estimates and the feed characteristics. Only variables relevant to each animal category are used (e.g., estimates for feedlot animals do not require the NE_l factor). All net energy equations are provided in IPCC (2000).

$$GE = [((NE_m + NE_{mobilized} + NE_a + NE_l + NE_p) / \{NE_{ma}/DE\}) + (NE_g / \{NE_{ga}/DE\})] / (DE / 100)$$

Where,

GE = gross energy (MJ/day)
NE_m = net energy required by the animal for maintenance (MJ/day)
$NE_{mobilized}$ = net energy due to weight loss (mobilized) (MJ/day)
NE_a = net energy for animal activity (MJ/day)
NE_l = net energy for lactation (MJ/day)
NE_p = net energy required for pregnancy (MJ/day)
$\{NE_{ma}/DE\}$ = ratio of net energy available in a diet for maintenance to digestible energy consumed
NE_g = net energy needed for growth (MJ/day)
$\{NE_{ga}/DE\}$ = ratio of net energy available for growth in a diet to digestible energy consumed
DE = digestible energy expressed as a percentage of gross energy (percent)

Step 3b: Emission Factor

The emissions factor (DayEmit) was determined using the GE value and the CH_4 conversion factor (Y_m) for each category. This is shown in the following equation:

$$DayEmit = [GE \times Y_m] / [55.65 \text{ MJ/kg } CH_4]$$

Where,

DayEmit = emission factor (kg CH_4/head/day)
GE = gross energy intake (MJ/head/day)
Y_m = CH_4 conversion rate which is the fraction of gross energy in feed converted to CH_4

The daily emission factors were estimated for each animal type, weight, and region.

Step 3c: Estimate Total Emissions

Emissions were summed for each month and for each population category using the daily emission factor for a representative animal and the number of animals in the category. The following equation was used:

$$\text{Emissions} = \text{DayEmit} \times \text{Days/Month} \times \text{SubPop}$$

Where,

DayEmit = the emission factor for the subcategory (kg CH_4/head/day)
Days/Month = the number of days in the month
SubPop = the number of animals in the subcategory during the month

This process was repeated for each month, and the totals for each subcategory were summed to achieve an emissions estimate for the entire year. The estimates for each of the 10 subcategories of cattle are listed in Appendix Table A-12. The emissions for each subcategory were then summed to estimate total emissions from beef cattle and dairy cattle for the entire year. The cattle emissions calculation model estimates emissions on a regional scale. Individual State-level estimates were developed from these regional estimates using the proportion of each cattle population subcategory in the State relative to the population in the region.

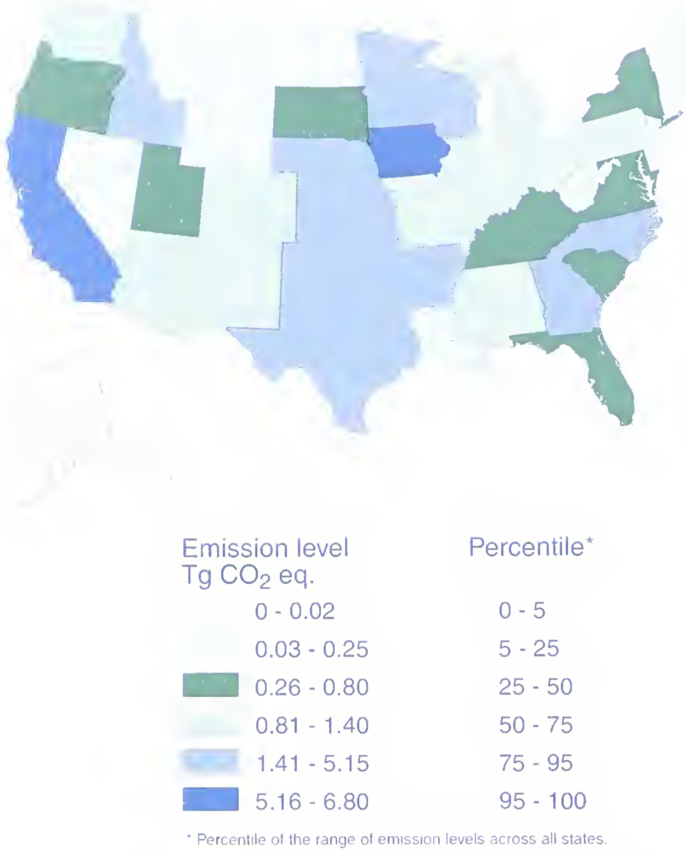
2.5.3 Emission Estimates From Other Livestock

All livestock population data, except for horses, were taken from USDA NASS reports (Hogs and Pigs: 2002, 2001, 2000, 1999, 1998, 1994; Sheep and Goats: 2002, 2001, 2000, 1999, 1994). Appendix Table A-2 shows the population data for all livestock that were used for estimating all livestock-related emissions. For each animal category, the USDA publishes monthly, annual, and multi-year livestock population and production estimates. Multi-year reports include revisions to earlier published data. Recent reports were obtained from the USDA Economics and Statistics System, while historical data were downloaded from USDA NASS. The Food and Agriculture Organization (FAO) of the United Nations publishes horse population data. These data were accessed from the FAOSTAT database (FAO 2002). National-level emission calculations for other livestock were developed from national population totals. State-level emissions were developed from these national totals based on the proportion of livestock population in each State relative to the national total population for the particular livestock category. Appendix Table A-13 shows the emission factors used for these other livestock.

2.6 Uncertainty in Estimating CH_4 Emissions from Enteric Fermentation

The following discussion of uncertainty in the enteric fermentation estimates is modified from that provided in the U.S. GHG Inventory and reproduced here with permission from EPA. Emission factors and animal population data are the primary sources of uncertainty in estimating CH_4 emissions from enteric fermentation. The estimation relies on a modeling approach that is sensitive to the accuracy of a number of input variables. The model estimates emission factors for the major animal types and diets, generating estimates for dairy and beef cows, dairy and beef replacements, beef stockers, and feedlot animals based on estimated energy requirements and diet characterizations. The model also estimates the movement of animal cohorts through monthly age and weight classes by animal type. Several inputs affect

Map 2-2
GHG emissions from livestock waste in 2001



the accuracy of this approach, including estimates of births by month, weight gain of animals by age class, and placement of animals into feedlots based on placement statistics and slaughter weight data. The model captures differences in values for Y_m and DE, reflecting diet characterizations assumed for each cattle group, within each region of the country. These values assume general diet characteristics within each region, thus local variation in feed characteristics are not captured.

2.7 Livestock Waste

GHG emissions from livestock waste come from several managed and unmanaged sources. Managed sources include CH_4 and N_2O from livestock waste storage and treatment and CH_4 emissions from the daily spread of livestock waste. Unmanaged sources include direct and indirect emissions of N_2O and emission of CH_4 from manure deposited on pasture, range, and paddock.[8] Emissions from these sources are discussed below, with estimates disaggregated by livestock category and waste management system where possible.

N_2O was the predominant greenhouse gas emitted from livestock waste in 2001, accounting for 66 percent of all emissions from this source (Table 2-4). The remaining 34 percent of GHG emissions from livestock waste was CH_4. In aggregate, N_2O from managed sources was lower than N_2O from unmanaged sources in 2001 and has been since 1990. This is the general pattern for all livestock categories with some exceptions. For example, 99 percent of N_2O from poultry waste was from managed sources in 2001 (Table 2-5).

N_2O emissions from unmanaged livestock waste totaled 59 Tg CO_2 eq. in 2001 (Table 2-4),

[8] Manure deposited on pasture, range, or paddock produces little CH_4 due to predominant aerobic conditions.

Table 2-4 GHG emissions from livestock waste, 1990-2001

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
						Tg CO_2 eq.						
Nitrous oxide	75.79	76.57	77.63	78.74	80.30	80.85	81.16	79.52	78.32	77.70	77.33	76.92
Managed	16.18	16.69	16.46	16.89	16.90	16.55	16.97	17.28	17.33	17.35	17.90	18.00
Unmanaged direct	40.42	40.60	41.47	41.94	42.99	43.59	43.52	42.19	41.35	40.91	40.30	39.95
Unmanaged indirect, volatilization	4.04	4.06	4.15	4.19	4.30	4.36	4.35	4.22	4.13	4.09	4.03	3.99
Unmanaged indirect, leaching & run-off	15.16	15.22	15.55	15.73	16.12	16.35	16.32	15.82	15.51	15.34	15.11	14.98
Methane[1]	31.34	33.26	32.19	32.99	35.52	36.25	34.95	36.63	39.10	38.98	38.35	39.01
Total	107.13	109.82	109.82	111.73	115.82	117.09	116.11	116.14	117.42	116.68	115.68	115.93

[1] Includes CH_4 from managed sources and from manure deposited on pasture, range, or paddock. Manure deposited on pasture, range, or paddock produces little CH_4 due to predominantly aerobic conditions.

including direct and indirect sources. Most N_2O from unmanaged livestock waste results from direct deposition of waste on pasture, range, and paddock. Beef cattle are responsible for the highest proportion of direct N_2O emissions from unmanaged waste (Table 2-5). Data are not available to estimate the indirect sources—leaching/run-off and volatilization—by livestock category. However, in general, leaching and run-off contribute more to indirect emissions of N_2O than volatilization (Table 2-4).

CH_4 emissions from livestock waste totaled 39 Tg CO_2 eq. in 2001 (Table 2-5). CH_4 from swine waste contributed the most to this total (17 Tg CO_2 eq.), while CH_4 from dairy cattle waste contributed the second largest portion (15 Tg CO_2 eq.) (Table 2-5). Beef cattle were responsible for only 3 Tg CO_2 eq. of CH_4 from waste and poultry for just under this amount. All other livestock types caused less than 1 Tg CO_2 eq. of CH_4 emissions each.

N_2O emissions from managed livestock waste totaled 18 Tg CO_2 eq. in 2001 (Table 2-4). Poultry waste contributed the most to N_2O emissions from managed waste sources (7.32 Tg CO_2 eq.) and N_2O emissions from managed beef cattle waste were slightly lower (6.10 Tg CO_2 eq.) (Table 2-5).

The remainder of this section discusses GHG emissions from managed and unmanaged livestock waste in aggregate, focusing on emissions by livestock type. Livestock-specific

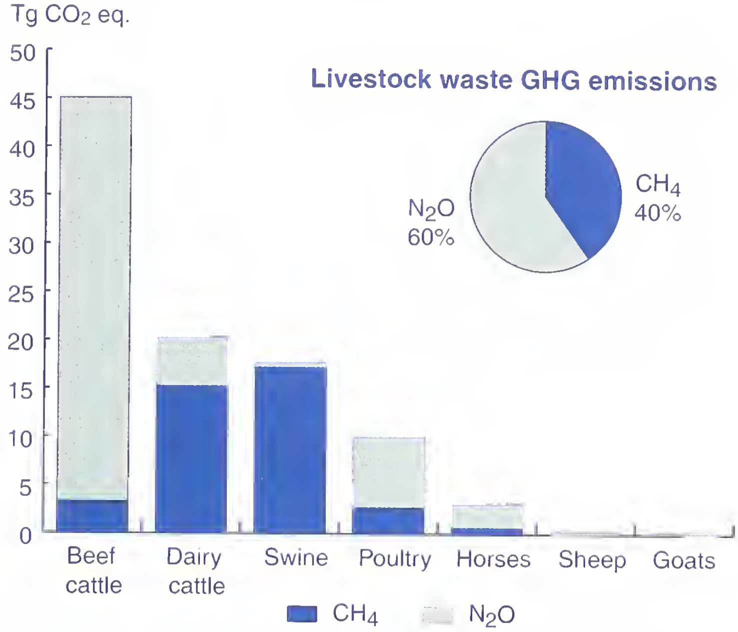
Table 2-5 GHG emissions from livestock waste by livestock category, 1990-2001, excluding indirect emissions from unmanaged waste

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
						Tg CO_2 eq.						
Beef cattle												
N_2O unmanaged	35.22	35.40	36.31	36.93	38.15	38.92	39.03	37.85	37.01	36.67	36.05	35.71
N_2O managed	4.91	5.36	5.02	5.36	5.27	5.29	5.10	5.40	5.51	5.54	5.89	6.10
CH_4[1]	3.37	3.35	3.38	3.39	3.45	3.45	3.45	3.38	3.32	3.32	3.29	3.26
Dairy cattle												
N_2O unmanaged	1.75	1.72	1.68	1.61	1.52	1.45	1.37	1.29	1.25	1.22	1.20	1.17
N_2O managed	4.30	4.23	4.18	4.16	4.12	4.11	4.04	3.99	3.93	3.95	3.95	3.92
CH_4[1]	11.47	12.31	12.08	11.91	13.11	13.47	12.87	13.47	13.94	14.75	14.63	15.20
Swine												
N_2O unmanaged	0.49	0.50	0.51	0.45	0.40	0.33	0.26	0.21	0.21	0.20	0.20	0.20
N_2O managed	0.36	0.38	0.40	0.40	0.42	0.42	0.40	0.42	0.45	0.43	0.42	0.42
CH_4[1]	13.11	14.19	13.42	14.29	15.56	16.03	15.33	16.43	18.42	17.64	17.13	17.17
Poultry												
N_2O unmanaged	0.07	0.08	0.08	0.08	0.08	0.08	0.09	0.09	0.09	0.09	0.09	0.09
N_2O managed	6.34	6.46	6.60	6.71	6.84	6.49	7.18	7.23	7.19	7.18	7.38	7.32
CH_4[1]	2.69	2.71	2.63	2.72	2.71	2.61	2.62	2.66	2.74	2.61	2.63	2.70
Horses												
N_2O unmanaged	2.24	2.25	2.26	2.27	2.26	2.27	2.27	2.28	2.31	2.28	2.31	2.34
N_2O managed	0.19	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20	0.20
CH_4[1]	0.60	0.61	0.61	0.61	0.61	0.61	0.61	0.62	0.62	0.62	0.62	0.63
Sheep												
N_2O unmanaged	0.41	0.41	0.39	0.37	0.36	0.33	0.31	0.29	0.29	0.26	0.26	0.25
N_2O managed	0.04	0.04	0.04	0.04	0.04	0.03	0.03	0.03	0.03	0.03	0.03	0.03
CH_4[1]	0.06	0.06	0.06	0.06	0.06	0.05	0.05	0.04	0.04	0.04	0.04	0.04
Goats												
N_2O unmanaged	0.24	0.24	0.24	0.23	0.22	0.21	0.20	0.19	0.19	0.19	0.19	0.19
N_2O managed	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02
CH_4[1]	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.01	0.01	0.01	0.01	0.01

[1] Includes CH_4 from managed sources and from manure deposited on pasture, range, or paddock. Manure deposited on pasture, range, or paddock produces little C114 due to predominantly aerobic conditions.

emissions are associated with common methods for handling waste either in a managed system or by unmanaged means. EPA, in preparing the official U.S. GHG Inventory, used several sources to assess the general fate of waste for various livestock operations based on their location and size, including input from field personnel in the USDA Natural Resources Conservation Service (NRCS) and industry experts, and data from the USDA National Agricultural Statistics Service (NASS). Trends in use of waste storage systems for different livestock types are provided below with corresponding estimates of GHG emissions from each. All N_2O estimates presented by livestock category include emissions from managed waste and direct emissions from unmanaged waste (i.e., Figure 2-1, Figure 2-2, Figure 2-3, Figure 2-4, Figure 2-5, Figure 2-6). These figures do not include indirect emissions from unmanaged manure since estimates by livestock type are not available.

Figure 2-2
U.S. Greenhouse gas emissions from livestock waste by livestock type, 2001



While beef cattle waste was typically deposited or stored either on pastures or dry lots, or collected through liquid/slurry systems, GHG emissions from waste in pastures was by far the largest source from beef cattle (Figure 2-3). N_2O emissions from waste in dry lots were also substantial, while those from liquid slurry systems were quite small. The large populations of U.S. beef cattle, coupled with the widespread use of pasture by the cow-calf and stocker sectors and dry lot systems for beef cattle in feedlots, drove this trend.

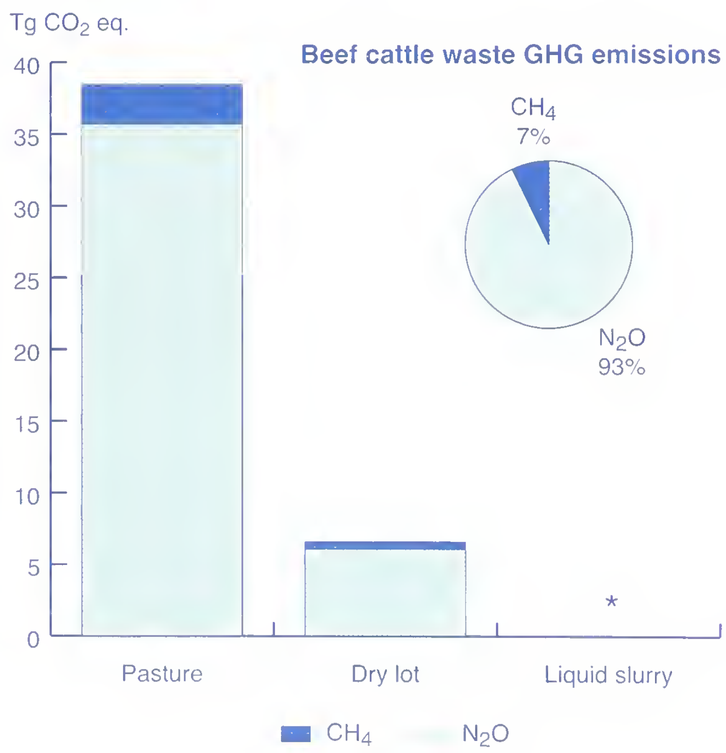
Medium (200-700 head) and large (>700 head) dairy operations typically managed waste with anaerobic lagoons or liquid/slurry systems, while liquid/slurry and solid storage systems were prevalent among smaller operations. In addition, daily spread, pasture, range, or paddock, and dry lot systems were used. In 2001, anaerobic lagoons contributed the most to overall greenhouse gas emissions from dairy cattle waste (Figure 2-4). Solid storage and liquid slurry systems were second in magnitude, while daily spreading of waste and deposition in dry lots were smaller sources. Waste from dairy cattle in pastures and waste stored in deep pits contributed relatively little to overall GHG emissions. This may be related to a trend in the dairy industry toward using large confined operations that feed with total mixed rations (TMR),

instead of grazing in pasture, range, and paddock. These large operations tend to use liquid manure management systems.

Medium (200-2,000 head) and large (>2,000 head) swine operations typically use deep pits, liquid/slurry systems, or anaerobic lagoons to store waste, while it is believed that small operations mainly let their swine graze in pasture, range, or paddock. In 2001, the majority of emissions from swine were from waste stored in deep pits (Figure 2-5). Nearly as large were emissions from anaerobic lagoons. Swine waste managed in liquid slurry systems contributed intermediate levels of GHG emissions, while waste in solid storage and deposited in pastures caused relatively small GHG emissions.

Poultry waste is typically deposited in shallow-pit flush houses coupled with anaerobic lagoons, high-rise houses without bedding, high-rise houses with bedding, or on pasture, range, or paddock. In 2001, the largest source of GHG emissions from poultry waste was from poultry houses where bedding is applied (Figure 2-6). Emissions from poultry houses without bedding and from anaerobic lagoons were smaller and were largely CH_4; emissions from poultry in pasture, range, and paddock systems were minimal (~0.01 Tg CO_2 eq.).

Figure 2-3
U.S. Greenhouse gas emissions from beef cattle waste, 2001

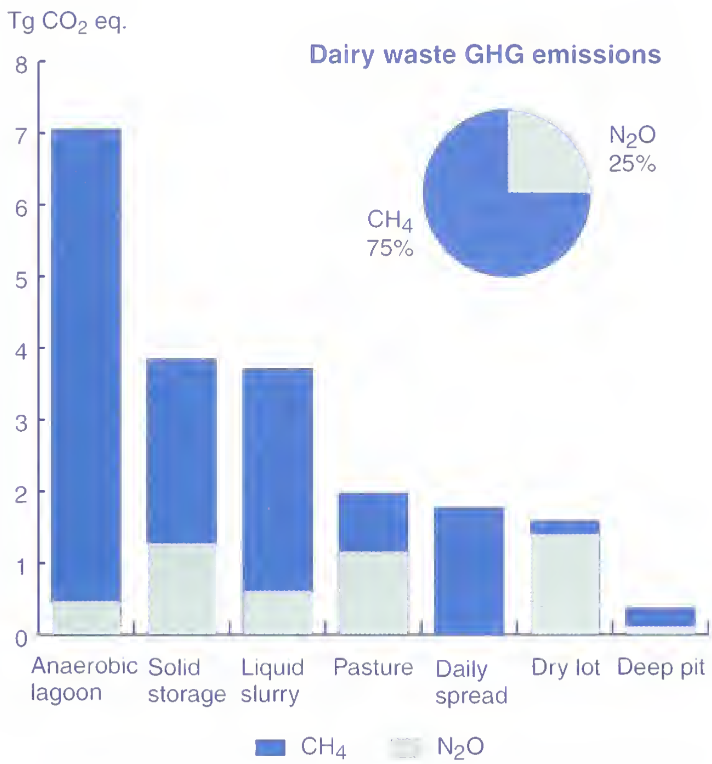


State-level GHG emissions estimates for livestock waste were developed based on the national methodology for all but one source. N_2O emissions from unmanaged livestock waste deposited on pasture, range, and paddock could only be presented at the national level in this report. Therefore, State-level estimates do not include N_2O emissions from manure deposited in pasture, range, and paddock, which are considerable for some livestock types, namely beef cattle.

State-level GHG emissions from managed livestock waste varied across States in 2001, with a small number of States responsible for the larger contributions to national GHG emissions. California and

Iowa had the largest GHG emissions from managed livestock waste (7 and 6 Tg CO_2 eq., respectively) (Appendix Table A-14, Appendix Table A-15, and Map 2-2). In California, GHG emissions from managed livestock waste were largely from dairy cattle, while in Iowa, they were largely from swine (Appendix Table A-14 and Appendix Table A-15). North Carolina and Texas also had large GHG emissions from managed livestock waste (5 and 4 Tg CO_2 eq., respectively). In North Carolina this was primarily from swine. In Texas, however, most emissions were from both beef and dairy cattle waste, with a smaller portion from swine.

Figure 2-4
U.S. Greenhouse gas emissions from dairy cattle manure, 2001



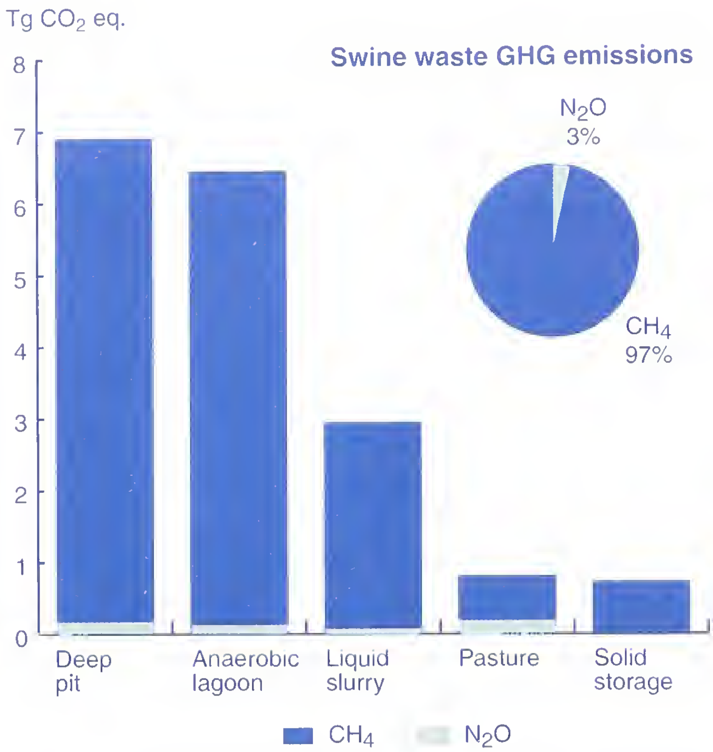
Estimated national emissions of CH_4 and N_2O from livestock waste have increased over the last 11 years (Table 2-4). N_2O emissions reached a peak of 81 Tg CO_2 eq. in 1996 or 8 percent higher than 1990 levels. N_2O emissions have decreased since 1996 and in 2001 were only 1 percent higher than 1990 levels. CH_4 emissions have increased consistently over the same time period, with 2001 levels 24 percent higher than in 1990, an increase of 8 Tg CO_2 eq. In total, emissions from livestock waste have increased by 8 percent since 1990.

2.8 Methods for Estimating CH_4 and N_2O Emissions from Livestock Waste

This section describes how CH_4 and N_2O emissions from livestock waste were calculated in the U.S. GHG Inventory (2003) and disaggregated for this inventory report. The U.S. GHG Inventory reports N_2O emission from livestock waste on pasture, range, and paddock separately from GHG emissions from treated or stored livestock waste, distinguishing between "managed" and "unmanaged" waste. This inventory reports direct and indirect emissions of N_2O from waste on pasture, range, and paddock with other managed waste sources.

EPA provided the USDA with State and national estimates of GHG emissions from managed

Figure 2-5
U.S. Greenhouse gas emissions from swine waste, 2001



livestock waste and national estimates of emissions from unmanaged waste. The estimates were prepared following a methodology developed by EPA and are described in Annexes M and N of the U.S. GHG Inventory. Annex M details the methodology for estimating GHG emissions from waste in managed systems. Annex N explains the methodology for estimating N_2O emissions from waste on pasture, range, and paddock. With permission from EPA, Annex M and the relevant portions of Annex N are reproduced below.

2.8.1 Annex M

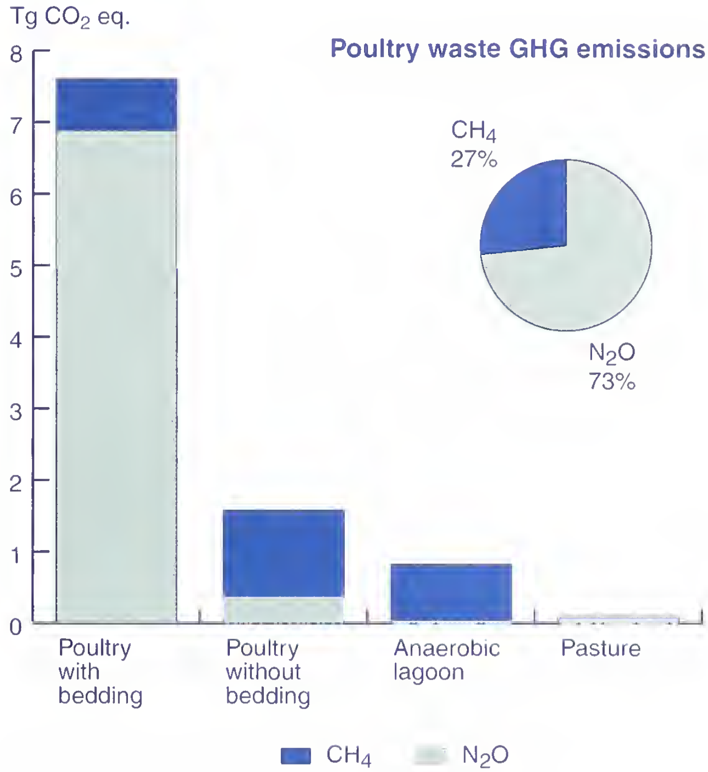
Step 1: Livestock Population Characterization Data

Annual animal population data for 1990 through 2001 for all livestock types, except horses and goats, were obtained from the USDA NASS (Cattle: 2002, 2001, 2000, 1999, 1998, 1995, 1994; Cattle on Feed: 2002, 2001, 2000, 1999, 1998, 1995, 1994; Hogs and Pigs: 2002, 2001, 2000, 1999; Chicken and Eggs: 2002, 2001, 2000; Poultry Production and Value: 2002, 2001, 2000; Sheep and Goats: 2002, 2001, 2000). Data for cattle and swine were downloaded from the USDA NASS Population Estimates Data Base (<http://www.usda.gov/nass/>) (USDA NASS 2001a). Horse population data were obtained from the FAOSTAT database (FAO 2002). Goat population data for 1992 and 1997 were obtained from the Census of Agriculture (USDA NASS 1999a). Information regarding poultry slaughter and mortality rates was obtained from USDA NRCS State personnel (Lange 2000). Livestock population data used to calculate CH_4 and N_2O emissions are in Appendix Table A-2.

Dairy Cattle: The total annual dairy cow and heifer State population data for 1990 through 2001 are provided in various USDA NASS reports (Cattle: 2002, 2001, 2000, 1999, 1995; Cattle on Feed: 2002, 2001, 2000). Data on annual dairy cow and heifer State population data used in the emissions calculations were downloaded from the USDA NASS Published Estimates Database for Cattle and Calves (USDA NASS 2001a). The specific data used to estimate dairy cattle populations are "cows that calved - milk" and "heifers 500+ lbs - milk repl."

Beef Cattle: The total annual beef cattle population data for each State for 1990 through 2001 are provided in various USDA NASS reports (Cattle: 2002, 2001, 2000, 1999, 1995; Cattle on Feed: 2002, 2001, 2000). Data used in the emissions calculations were downloaded from the USDA NASS Published Estimates Database for Cattle and Calves (USDA NASS 2001a). The specific data used to estimate beef cattle populations are: "cows that calved—beef," "heifers 500+ lbs—beef repl," "heifers 500+ lbs—other," and "steers 500+ lbs." Additional information regarding the percent of beef steer and heifers in feedlots was obtained from NASS contacts (Milton 2000).

Figure 2-6
U.S. Greenhouse gas emissions from poultry waste, 2001



For all beef cattle groups (cows, heifers, steer, bulls, and calves), the USDA data provide cattle inventories from January and July of each year. Cattle inventories change over the course of the year, sometimes significantly, as new calves are born and as fattened cattle are slaughtered; therefore, to develop the best estimate for the annual animal population, the average inventory of cattle by State was calculated. USDA provides January inventory data for each State; however, July inventory data is only presented as a total for the United States. In order to estimate average annual populations by State, a "scaling factor" was developed that adjusts the January State-level data to reflect July inventory changes. This factor equals the average of the U.S. January and July data divided by the January data. The scaling factor is derived for each cattle group and is then applied to the January State-level data to arrive at the State-level annual population estimates.

Swine: The total annual swine population data for each State for 1990 through 2001 are provided in various USDA NASS reports (Hogs and Pigs: 2002, 2001, 2000, 1998, 1994). The USDA source provides quarterly data for each swine subcategory: breeding, market under 60 pounds (less than 27 kg), market 60 to 119 pounds (27 to 54 kg), market 120 to 179 pounds (54 to 81 kg), and market 180 pounds and over (greater than 82 kg). The average of the quarterly

data was used in the emissions calculations. For States where only a December inventory is reported, the December data were used directly. Data used in the emissions calculations were downloaded from the USDA NASS Published Estimates Database for Hogs and Pigs (USDA NASS 2001a).

Sheep: The total annual sheep population data for each State for 1990 through 2001 were obtained from USDA NASS reports (Sheep and Goats: 2002, 2001, 2000, 1999, 1994). Population data for lambs and sheep on feed are not available after 1993. The number of lambs and sheep on feed for 1994 through 2001 were calculated using the average percent of lambs and sheep on feed from 1990 through 1993. In addition, all of the sheep and lambs "on feed" are not necessarily in feedlots; they may be on pasture/crop residue supplemented by feed. Data for those animals on feed that are on feedlots versus pasture/crop residue were provided only for lambs in 1993. To calculate the populations of sheep and lambs on feedlots for all years, it was assumed that the percentage of sheep and lambs on feed that are on feedlots versus pasture/crop residue is the same as that for lambs in 1993 (Anderson 2000).

Goats: Annual goat population data by State were available for only 1992 and 1997 (USDA NASS 1999a). The data for 1992 were used for 1990 through 1992 and the data for 1997 were used for 1997 through 2001. Data for 1993 through 1996 were extrapolated using the 1992 and 1997 data.

Poultry: Annual poultry population data by State for the various animal categories (hens 1 year and older, total pullets, other chickens, broilers, and turkeys) were obtained from USDA NASS (Chicken and Eggs: 2002, 2001, 2000, 1998; Poultry Production and Value: 2002, 2001, 1999, 1995). The annual population data for boilers and turkeys were adjusted for slaughter and mortality rate (Lange 2000).

Horses: The FAO publishes annual horse population data, which were accessed from the FAOSTAT database (FAO 2002).

Step 2: Waste Characteristics Data

CH_4 and N_2O emissions calculations are based on the following animal characteristics for each relevant livestock population:

- Volatile solids excretion rate (VS)
- Maximum CH_4 producing capacity (B_o) for U.S. animal waste
- Nitrogen excretion rate (Nex)
- Typical animal mass (TAM)

Appendix Table A-16 presents a summary of the waste characteristics used in the emissions estimates. Published sources were reviewed for U.S.-specific livestock waste characterization data that would be consistent with the animal population data discussed in Step 1. The USDA's

National Engineering Handbook, Agricultural Waste Management Field Handbook (USDA NRCS 1996) is one of the primary sources of waste characteristics. In some cases, data from the American Society of Agricultural Engineers, Standard D384.1 (ASAE 1999) were used to supplement the USDA data. The volatile solids and nitrogen excretion data for breeding swine are a combination of the types of animals that make up this animal group, namely gestating and farrowing swine and boars. It is assumed that a group of breeding swine is typically broken out as 80 percent gestating sows, 15 percent farrowing swine, and 5 percent boars (Safley 2000).

The method for calculating volatile solids production from beef and dairy cows, heifers, and steers is based on the relationship between animal diet and energy utilization, which is modeled in the enteric fermentation portion of the inventory. Volatile solids content of manure equals the fraction of the diet consumed by cattle that is not digested and thus excreted as fecal material which, when combined with urinary excretions, constitutes manure. The enteric fermentation model requires the estimation of gross energy intake and its fractional digestibility, digestible energy, in the process of estimating enteric CH_4 emissions (see section 2.5.2 for details on the enteric energy model). These two inputs were used to calculate the indigestible energy per animal unit as gross energy minus digestible energy plus an additional 2 percent of gross energy for urinary energy excretion per animal unit. This was then converted to volatile solids production per animal unit using the typical conversion of dietary gross energy to dry organic matter of 20.1 MJ/kg (Garrett and Johnson, 1983). The equation used for calculating volatile solids is as follows:

$$VS_{production}(kg) = \frac{[GE - DE + (0.02 \times GE)]}{20.1 MJkg^{-1}}$$

Where:

GE= gross energy intake (MJ)
DE= digestible energy (MJ)

This equation was used to calculate volatile solids rates for each region, cattle type, and year, with State-specific volatile solids excretion rates assigned based on the region where the State is located (Peterson et al., 2002). VS rates for cattle, which are outputs from enteric fermentation model, reflect changes in the time series due to underlying data for DE and lactation rates. For all other species, VS rates (kg VS/1000kg animal mass/day) are constant for the time series. Future work will consider updates to these VS rates. Appendix Table A-17 presents the State-specific volatile solids production rates used for 2001.

Step 3: Waste Management System Usage Data

Estimates were made of the distribution of wastes by management system and animal type using the following sources of information:

- State contacts to estimate the breakout of dairy cows on pasture, range, or paddock, and

the percent of waste managed by daily spread systems (Deal 2000, Johnson 2000, Miller 2000, Stettler 2000, Sweeten 2000, Wright 2000);

- Data collected for EPA's Office of Water, including site visits, to medium and large beef feedlot, dairy, swine, and poultry operations (EPA 2001a);
- Contacts with the national USDA office to estimate the percent of beef steer and heifers on feedlots (Milton 2000);
- Survey data collected by USDA (APHIS NAHMS Special request 1998, 2000) and re-aggregated by farm size and geographic location, used for small operations;
- Survey data collected by the United Egg Producers (UEP 1999) and USDA (APHIS NAHMS 2000) and previous EPA estimates (EPA 1992a) of waste distribution for layers;
- Survey data collected by Cornell University on dairy manure management operations in New York (Poe et al. 1999); and
- Previous EPA estimates of waste distribution for sheep, goat, and horse operations (EPA 1992a).

Beef Feedlots: Based on EPA site visits and State contacts, beef feedlot manure is almost exclusively managed in drylots. Therefore, 100 percent of the manure excreted at beef feedlots is expected to be deposited in drylots and generate emissions. In addition, a portion of the manure that is deposited in the drylot will run off the drylot during rain events and be captured in a waste storage pond. An estimate of the runoff has been made by EPA's Office of Water for various geographic regions of the United States. These runoff numbers were used to estimate emissions from runoff storage ponds located at beef feedlots (EPA 2001a).

Dairy Cows: Based on EPA site visits and State contacts, manure from dairy cows at medium (200 through 700 head) and large (greater than 700 head) operations is managed using either flush systems or scrape/slurry systems. In addition, they may have a solids separator in place prior to their storage component. Estimates of the percent of farms that use each type of system (by geographic region) were developed by EPA's Office of Water, and were used to estimate the percent of wastes managed in lagoons (flush systems), liquid/slurry systems (scrape systems), and solid storage (separated solids) (EPA 2001a). Manure management system data for small (fewer than 200 head) dairies were obtained from USDA (APHIS NAHMS special request 2000). These operations are more likely to use liquid/slurry and solid storage management systems than anaerobic lagoon systems. The reported manure management systems were deep pit, liquid/slurry (also includes slurry tank, slurry earth-basin, and aerated lagoon), anaerobic lagoon, and solid storage (also includes manure pack, outside storage, and inside storage).

The percent of wastes by system was estimated using the USDA data broken out by geographic region and farm size. Farm-size distribution data reported in the 1992 and 1997 Census of Agriculture (USDA NASS Census 1999) were used to determine the percentage of all dairies using the various manure management systems. Due to a lack of additional data for other years,

it was assumed that the data provided for 1992 were the same as those for 1990 and 1991, and data provided for 1997 were the same as that for 1998, 1999, 2000, and 2001. Data for 1993 through 1996 were interpolated using the 1992 and 1997 data.

Data regarding the use of daily spread and pasture, range, or paddock systems for dairy cattle were obtained from personal communications with personnel from several organizations. These organizations include State NRCS offices, State extension services, State universities, USDA NASS, and other experts (Deal 2000, Johnson 2000, Miller 2000, Stettler 2000, Sweeten 2000, and Wright 2000). Contacts at Cornell University provided survey data on dairy manure management practices in New York (Poe et al. 1999). Census of Agriculture population data for 1992 and 1997 (USDA NASS Census 1999) were used in conjunction with the State data obtained from personal communications to determine regional percentages of total dairy cattle and dairy wastes that are managed using these systems. These percentages were applied to the total annual dairy cow and heifer State population data for 1990 through 2001 which were obtained from the USDA NASS (Cattle: 2002, 2001, 2000, 1999, 1995; Cattle on Feed: 2002, 2001, 2000).

Of the dairies using systems other than daily spread and pasture, range, or paddock systems, some dairies reported using more than one type of manure management system. Therefore, the total percent of systems reported by USDA for a region and farm size is greater than 100 percent. Typically, this means that some of the manure at a dairy is handled in one system (e.g., a lagoon), and some of the manure is handled in another system (e.g., drylot). However, it is unlikely that the same manure is moved from one system to another. Therefore, to avoid double counting emissions, the reported percentages of systems in use were adjusted to equal a total of 100 percent, using the same distribution of systems. For example, if USDA reported that 65 percent of dairies use deep pits to manage manure and 55 percent of dairies use anaerobic lagoons to manage manure, it was assumed that 54 percent (i.e., 65 percent divided by 120 percent) of the manure is managed with deep pits and 46 percent (i.e., 55 percent divided by 120 percent) of the manure is managed with anaerobic lagoons (ERG 2000a).

Dairy Heifers: The percent of dairy heifer operations that are pasture, range, or paddock or that operate as daily spread was estimated using the same approach as dairy cows. Similar to beef cattle, dairy heifers are housed on drylots when not pasture based. Based on data from EPA's Office of Water (EPA 2001a), it was assumed that 100 percent of the manure excreted by dairy heifers is deposited in drylots and generates emissions. Estimates of runoff have been made by EPA's Office of Water for various geographic regions of the U.S. (EPA 2001a).[9]

Swine: Based on data collected during site visits for EPA's Office of Water (ERG 2000a), manure from swine at large (greater than 2,000 head) and medium (200 through 2,000 head)

[9] The amount of nitrogen and VS managed in runoff collection ponds is estimated from nitrogen and VS in annual runoff. The daily runoff volume is estimated as the 6-month runoff volume divided by 180 days plus the 25-year/24-hour storm runoff divided by 365 days (this overestimates the runoff volume since the 25-year storm does not happen every year). The amount of solids in the runoff volume is assumed to be 1.5 percent of the runoff mass; solids are calculated as the runoff volume (ft^3) times 62.4 lb/cf (density of water) times 0.015 and are assumed to have the same characteristics as manure to estimate nitrogen and VS content.

operations are primarily managed using deep pit systems, liquid/slurry systems, or anaerobic lagoons. Manure management system data were obtained from USDA (APHIS NAHMS Special request 1998). It was assumed those operations with less than 200 head use pasture, range, or paddock systems. The percent of waste by system was estimated using the USDA data broken out by geographic region and farm size. Farm-size distribution data reported in the 1992 and 1997 Census of Agriculture (USDA NASS Census 1999) were used to determine the percentage of all swine utilizing the various manure management systems. The reported manure management systems were deep pit, liquid/slurry (also includes above- and below-ground slurry), anaerobic lagoon, and solid storage (also includes solids separated from liquids).

Some swine operations reported using more than one management system; therefore, the total percent of systems reported by USDA for a region and farm size is greater than 100 percent. Typically, this means that some of the manure at a swine operation is handled in one system (e.g., liquid system), and some of the manure is handled in another system (e.g., dry system). However, it is unlikely that the same manure is moved from one system to another. Therefore, to avoid double counting emissions, the reported percentages of systems in use were adjusted to equal a total of 100 percent, using the same distribution of systems, as explained under "Dairy Cows."

Sheep: It was assumed that all sheep wastes not deposited on feedlots were deposited on pasture, range, or paddock lands (Anderson 2000).

Goats/Horses: Estimates of manure management distribution were obtained from EPA's previous estimates (EPA 1992a).

Poultry – Layers: Waste management system data for layers for 1990 were obtained from Appendix H of Global CH_4 Emissions from Livestock and Poultry Manure (EPA 1992a). The percentage of layer operations using a shallow pit flush house with anaerobic lagoon or high-rise house without bedding was obtained for 1999 from United Egg Producers, voluntary survey, 1999 (UEP 1999). These data were augmented for key poultry States (Alabama, Arkansas, California, Florida, Georgia, Iowa, Indiana, Minnesota, Missouri, North Carolina, Nebraska, Ohio, Pennsylvania, Texas, and Washington) with USDA data (AHPIS NAHMS 2000). It was assumed that the change in system usage between 1990 and 1999 is proportionally distributed among those years of the inventory. It was assumed that system usage in 2000 and 2001 was equal to that estimated for 1999. It was also assumed that 1 percent of poultry wastes are deposited on pasture, range, or paddock lands (EPA 1992a).

Poultry - Broilers/Turkeys: The percentage of turkeys and broilers on pasture or in high-rise houses without bedding was obtained from Global CH_4 Emissions from Livestock and Poultry Manure (EPA1992). It was assumed that 1 percent of poultry wastes are deposited in pastures, range, and paddocks (EPA 1992a).

Step 4: Emission Factor Calculations

CH_4 conversion factors (MCFs) and N_2O emission factors (EFs) used in the emission calculations were determined using the methodologies shown below:

Methane Conversion Factors (MCFs)

Good Practice Guidance and Uncertainty Management in National GHG Inventories (IPCC 2000) for anaerobic lagoon systems published default CH_4 conversion factors of 0 to 100 percent, which reflects the wide range in performance that may be achieved with these systems. There exist relatively few data points on which to determine country-specific MCFs for these systems. Therefore, a climate-based approach was identified to estimate MCFs for anaerobic lagoon and other liquid storage systems. The following approach was used to develop the MCFs for liquid systems, and is based on the van't Hoff-Arrhenius equation used to forecast performance of biological reactions. One practical way of estimating MCFs for liquid manure handling systems is based on the mean ambient temperature and the van't Hoff-Arrhenius equation with a base temperature of 30°C, as shown in the following equation (Safley and Westerman 1990):

$$f = \exp\left[\frac{E(T_2 - T_1)}{RT_1T_2}\right]$$

Where:

T_1 = 303.16K

T_2 = ambient temperature (K) for climate zone (in this case, a weighted value for each State)

E = activation energy constant (15,175 cal/mol)

R = ideal gas constant (1.987 cal/K mol)

The factor "f" represents the proportion of volatile solids that are biologically available for conversion to CH_4 based on the temperature of the system. The temperature is assumed equal to the ambient temperature. For colder climates, a minimum temperature of 5°C was established for uncovered anaerobic lagoons and 7.5°C for other liquid manure handling systems. For those animal populations using liquid systems (i.e., dairy cow, dairy heifer, layers, beef on feedlots, and swine) monthly average State temperatures were based on the counties where the specific animal population resides (i.e., the temperatures were weighted based on the percent of animals located in each county). The average county and State temperature data were obtained from the National Climate Data Center (NOAA 2001), and the county population data were based on 1992 and 1997 Census data (USDA NASS Census 1999). County population data for 1990 and 1991 were assumed to be the same as 1992; county population data for 1998 through 2001 were assumed to be the same as 1997; and county population data for 1993 through 1996 were extrapolated based on 1992 and 1997 data.

Annual MCFs for liquid systems are calculated as follows for each animal type, State, and year of the inventory:

(1) Monthly temperatures are calculated using county-level temperature and population data. The weighted-average temperature for a State is calculated using the population estimates and average monthly temperature in each county.
(2) Monthly temperatures are used to calculate a monthly van't Hoff-Arrhenius "f" factor, using the equation presented above. A minimum temperature of 5°C is used for anaerobic lagoons and 7.5°C is used for liquid/slurry and deep-pit systems.
(3) Monthly production of volatile solids that are added to the system is estimated based on the number of animals present and, for lagoon systems, adjusted for a management and design practices factor. This factor accounts for other mechanisms by which volatile solids are removed from the management system prior to conversion to CH_4, such as solids being removed from the system for application to cropland. This factor, equal to 0.8, has been estimated using currently available CH_4 measurement data from anaerobic lagoon systems in the United States (ERG 2001).
(4) The amount of volatile solids available for conversion to CH_4 is assumed to be equal to the amount of volatile solids produced during the month (from Step 3). For anaerobic lagoons, the amount of volatile solids available also includes volatile solids that may remain in the system from previous months.
(5) The amount of volatile solids consumed during the month is equal to the amount available for conversion multiplied by the "f" factor.
(6) For anaerobic lagoons, the amount of volatile solids carried over from one month to the next is equal to the amount available for conversion minus the amount consumed.
(7) The estimated amount of CH_4 generated during the month is equal to the monthly volatile solids consumed multiplied by the maximum CH_4 potential of the waste (B_o).
(8) The annual MCF is then calculated as:

$$MCF_{annual} = \frac{CH_4\ generated_{annual}}{(VS\ generated_{annual} \times B_o)}$$

Where:

MCF_{annual} = CH_4 conversion factor
$VS\ generated_{annual}$ = Volatile solids excretion rate
B_o = Maximum CH_4 producing potential of the waste

In order to account for the carry over of volatile solids from the year prior to the inventory year for which estimates are calculated, it is assumed in the MCF calculation for lagoons that a portion of the volatile solids from October, November, and December of the year prior to the inventory year are available in the lagoon system starting January of the inventory year. Following this procedure, the resulting MCF accounts for temperature variation throughout the year, residual volatile solids in a system (carry over), and management and design practices that may reduce the volatile solids available for conversion to CH_4. The base MCFs are shown in Appendix Table A-18 by State and waste management systems for which State factors were

used (liquid slurry, anaerobic lagoon, deep pit). These data are the average MCF for 2001 by State for all animal groups located in that State and are provided for illustrative purposes. However, in the calculation of CH_4 emissions, specific MCFs for each animal type in the State are used. For other waste management systems, default IPCC emission factors were used (Appendix Table A-19).

Nitrous Oxide Emission Factors (EFs)

N_2O emission factors for all manure management systems were set equal to the default IPCC factors (IPCC 2000) (Appendix Table A-19).

Step 5: Weighted Emission Factors

For beef cattle, dairy cattle, swine, and poultry, the emission factors for both CH_4 and N_2O were weighted to incorporate the distribution of wastes by management system for each State. The following equation was used to determine the weighted MCF for a particular animal type in a particular State:

$$MCF_{animal,State} = \sum_{system}\left(MCF_{system,State} \times \% Manure_{animal,system,State}\right)$$

Where:

$MCF_{animal,\ State}$ = Weighted MCF for that animal group and State
$MCF_{system,\ State}$ = MCF for that system and State (see Step 4)
$\% Manure_{animal,\ system,\ State}$ = Percent of manure managed in the system for that animal group in that State (expressed as a decimal)

The weighted N_2O emission factor for a particular animal type in a particular State was determined as follows:

$$EF_{animal,State} = \sum_{system}\left(EF_{system} \times \% Manure_{animal,system,State}\right)$$

Where:

$EF_{animal,\ State}$ = Weighted emission factor for that animal group and State
EF_{system} = Emission factor for that system (see Step 4)
$\% Manure_{animal,\ system,\ State}$ = Percent of manure managed in the system for that animal group in that State (expressed as a decimal)

Data for the calculated weighted factors for 1992 came from the 1992 Census of Agriculture (USDA NASS Census 1999), combined with assumptions on manure management system usage based on farm size, and were also used for 1990 and 1991. Data for the calculated weighted factors for 1997 came from the 1997 Census of Agriculture (USDA NASS Census 1999), combined with assumptions on manure management system usage based on farm size, and were also used for 1998, 1999, 2000, and 2001. Factors for 1993 through 1996 were calculated by interpolating between the two sets of factors. A summary of the weighted MCFs

used to calculate beef feedlot, dairy cow and heifer, swine, and poultry emissions for 2001 is presented in Appendix Table A-20.

Step 6: Methane and Nitrous Oxide Emission Calculations

For beef feedlot cattle, dairy cows, dairy heifers, swine, and poultry, CH_4 emissions were calculated for each animal group as follows:

$$Methane_{animal,group} = \sum_{State} \left(Population \times VS \times B_o \times MCF_{animal,State} \times 0.662 \right)$$

Where:

$Methane_{animal\,group}$ = CH_4 emissions for that animal group (kg CH_4/yr)
$Population$ = annual average State animal population for that animal group (head)
VS = total volatile solids produced annually per animal (kg/yr/head)
B_o = maximum CH_4 producing capacity per kilogram of VS (m^3 CH_4/kg VS)
$MCF_{animal,\,State}$ = weighted MCF for the animal group and State (see Step 5)
0.662 = conversion factor of m^3 CH_4 to kilograms CH_4 (kg CH_4 /m^3 CH_4)

CH_4 emissions from other animals (i.e., sheep, goats, and horses) were based on the 1990 CH_4 emissions estimated using the detailed method described in Anthropogenic Methane Emissions in the United States: Estimates for 1990, Report to Congress (EPA 1993b). This approach is based on animal-specific manure characteristics and management system data. This process was not repeated for subsequent years for these other animal types. Instead, national populations of each of the animal types were used to scale the 1990 emissions estimates to the period 1991 through 2001.

N_2O emissions were calculated for each animal group as follows:

$$Nitrous\ Oxide_{animal,group} = \sum_{State} \left(Population \times N_{ex} \times EF_{animal,State} \times {}^{44}\!/_{28} \right)$$

Where:

$Nitrous\ Oxide_{animal\,group}$ = N_2O emissions for that animal group (kg/yr)
$Population$ = annual average State animal population for that animal group (head)
N_{ex} = total Kjeldahl nitrogen excreted annually per animal (kg/yr/head)
$EF_{animal,\,State}$ = weighted N_2O emission factor for the animal group and State, kg N_2O-N/kg N excreted (see Step 5)
44/28 = conversion factor of N_2O-N to N_2O

2.8.2 Annex N (excerpts on emissions from livestock on pasture, range, and paddock)

Direct N_2O Emissions from Pasture, Range, and Paddock Livestock Manure
Estimates of N_2O emissions from this component were based on livestock manure that is not

managed in manure management systems, but instead is deposited directly on soils by animals in pasture, range, and paddock. The livestock included in this component were: dairy cattle, beef cattle, swine, sheep, goats, poultry, and horses.

Dairy Cattle: Information regarding dairy farm grazing was obtained from communications with personnel from State NRCS offices, State universities, and other experts (Poe et al. 1999, Deal 2000, Johnson 2000, Miller 2000, Stettler 2000, Sweeten 2000, Wright 2000). Because grazing operations are typically related to the number of animals on a farm, farm-size distribution data reported in the 1992 and 1997 Census of Agriculture (USDA NASS Census 1999) were used in conjunction with the State data obtained from personal communications to determine the percentage of total dairy cattle that graze. An overall percent of dairy waste that is deposited in pasture, range, and paddock was developed for each region of the United States. This percentage was applied to the total annual dairy cow and heifer State population data for 1990 through 2001, which were obtained from the USDA NASS (Cattle: 2002, 2001, 2000, 1999, 1995; Cattle on Feed: 2002, 2001, 2000).

Beef Cattle: To determine the population of beef cattle that are on pasture, range, and paddock, the following assumptions were made: 1) beef cows, bulls, and calves were not housed on feedlots; 2) a portion of heifers and steers were on feedlots; and 3) all beef cattle that were not housed on feedlots were located on pasture, range, and paddock (i.e., total population minus population on feedlots equals population of pasture, range, and paddock) (Milton 2000). Information regarding the percentage of heifers and steers on feedlots was obtained from USDA personnel (Milton 2000) and used in conjunction with population data from USDA NASS (Cattle: 2002, 2001, 2000, 1999, 1995; Cattle on Feed: 2002, 2001, 2000) to determine the population of steers and heifers on pasture, range, and paddock.

Swine: Based on the assumption that smaller facilities are less likely to utilize manure management systems, farm-size distribution data reported in the 1992 and 1997 Census of Agriculture (USDA NASS Census 1999) were used to determine the percentage of all swine whose manure is not managed (i.e., the percentage on pasture, range, and paddock). These percentages were applied to the average of the quarterly population data for swine published by USDA NASS (Hogs and Pigs: 2002, 2001, 2000, 1998, 1994) to determine the population of swine on pasture, range, and paddock.

Sheep: It was assumed that all sheep and lamb manure not deposited on feedlots was deposited on pasture, range, and paddock (Anderson 2000). Sheep population data were obtained from the USDA NASS (Sheep and goats: 2002, 2001, 2000, 1999, 1994). However, population data for lamb and sheep on feed were not available after 1993. The number of lamb and sheep on feed for 1994 through 2001 were calculated using the average of the percent of lamb and sheep on feed from 1990 through 1993. In addition, all of the sheep have been on pasture/crop residue supplemented by feed. Data for those feedlot animals versus pasture/crop residue were provided only for lamb in 1993. To calculate the populations of sheep and lamb on feedlots for

all years, it was assumed that the percentage of sheep and lamb on feedlots versus pasture/crop residue is the same as that for lambs in 1993 (Anderson 2000).

Goats: It was assumed that 92 percent of goat manure was deposited on pasture, range, and paddock (Safley et al. 1992). Annual goat population data by State were available for only 1992 and 1997 (USDA NASS 1999a). The data for 1992 were used for 1990 through 1992 and the data for 1997 were used for 1997 through 2001. Data for 1993 through 1996 were extrapolated using the 1992 and 1997 data.

Poultry: It was assumed that 1 percent of poultry manure was deposited on pasture, range, and paddock (Safley et al. 1992). Poultry population data were obtained from USDA NASS (Poultry Production and Value: 2002, 2001, 1999, 1995; Chicken and Eggs: 2000, 1998). The annual population data for boilers and turkeys were adjusted for slaughter and mortality rates (Lange 2000).

Horses: It was assumed that 92 percent of horse manure was deposited on pasture, range, and paddock (Safley et al. 1992). Horse population data were obtained from the FAOSTAT database (FAO 2002).

For each animal type, the population of animals within pasture, range, and paddock systems was multiplied by an average animal mass constant (Safley 2000, USDA NRCS 1998, ASAE 1999, USDA NRCS 1996) to derive total animal mass for each animal type. Total Kjeldahl nitrogen excreted per year was then calculated for each animal type using daily rates of nitrogen excretion per unit of animal mass (ASAE 1999, USDA NRCS 1996). Annual nitrogen excretion was then summed over all animal types to yield total nitrogen in pasture, range, and paddock manure (Appendix Table A-21).

Estimated Direct N_2O Emissions from Pasture, Range, and Paddock Livestock Manure
To estimate direct N_2O emissions from soils due to the deposition of pasture, range, and paddock manure, the total nitrogen excreted by these animals was multiplied by the IPCC default emission factor (0.02 kg N_2O-N/kg N excreted).

Estimated Indirect N_2O Emissions from Pasture, Range, and Paddock Livestock Manure
In this step, N_2O emissions were calculated for each of two parts (indirect N_2O emissions due to volatilization of applied nitrogen and indirect N_2O emissions due to leaching and runoff of applied nitrogen), which were then summed to yield total direct N2O emissions from managed soils.

Volatilization: The amount of manure nitrogen deposited in pasture, range, and paddock was multiplied by the IPCC default fraction of nitrogen that is assumed to volatilize to NH_3 and NO_x (20 percent for nitrogen in organic livestock manure). The total volatilized nitrogen was multiplied by the IPCC default emission factor of 0.01 kg N20- N/kg N (IPCC/UNEP/OECD/

IEA 1997).

Leaching and Runoff: The amount of manure nitrogen deposited on pasture, range, and paddock was multiplied by the IPCC default fraction of nitrogen that is assumed to leach and runoff (30 percent for all nitrogen). The total nitrogen was multiplied by the IPCC default emission factor of 0.025 kg N_20-N/kg N (IPCC/UNEP/OECD/IEA 1997).

2.9 Uncertainty in Estimating CH_4 and N_2O Emissions from Livestock Waste

The following discussion of uncertainty in estimating GHG emissions from livestock waste is modified from information provided in the U.S. GHG Inventory. The information is reproduced here with permission from EPA.

2.9.1 Managed Waste

Uncertainties derive from limited information on regional patterns in the use of manure management systems and CH_4 generating characteristics of each system. It is assumed that shifts in the swine and dairy sectors toward larger farms cause more manure to be managed in liquid manure management systems. Farm-size data from 1992 and 1997 are used to modify MCFs based on this assumption. However, the assumption of a direct relationship between farm size and liquid system usage may not apply in all cases and may vary based on geographic location. In addition, the CH_4 generating characteristics of manure management systems are based on relatively few laboratory and field measurements. Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories (IPCC 2000) published a default range of MCFs for anaerobic lagoon systems of 0 to 100 percent, reflecting the wide range in performance of these systems.

There are potential classification errors when naming manure management systems. For example, many livestock waste treatment systems classified as anaerobic lagoons are actually holding ponds, which may be organically overloaded, thus producing CH_4 at a different rate than assumed. In addition, the performance of manure management systems depends on how they are operated, which undoubtedly varies across facilities. An MCF based on optimized lagoon systems does not take into consideration the actual variation in performance across operational systems. Therefore, an MCF methodology was developed to better match observed system performance and account for the impact of temperature on system performance. The MCF methodology used in the inventory includes a factor to account for management and design practices that result in the loss of volatile solids from the management system. This factor, estimated with data from three systems, all in anaerobic lagoons in temperate climates, was applied broadly to systems across a range of management practices. Additional data are needed on animal waste lagoon systems across the country to verify and refine this methodology. Data are also needed on how lagoon temperatures relate to ambient air temperatures and whether the lower bound estimate of temperature used for lagoons and other liquid systems should be revised. The inventory relies on the IPCC MCF for poultry waste

management operations of 1.5 percent. This factor needs further evaluation to assess if poultry high-rise houses promote sufficient aerobic conditions to warrant a lower MCF.

The default N_2O emission factors published in Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories (IPCC 2000) were derived using limited information. The IPCC factors are global averages; U.S.-specific emission factors may be significantly different. Manure and urine in anaerobic lagoons and liquid/slurry management systems produce CH_4 at different rates, and would in all likelihood produce N_2O at different rates, although a single N_2O emission factor was used for both system types. In addition, there are little data available to determine the extent to which nitrification and denitrification occur in animal waste management systems. Ammonia concentrations that are present in poultry and swine systems suggest that N_2O emissions from these systems may be lower than predicted by the IPCC default factors. At this time, there are insufficient data available to develop U.S.-specific N_2O emission factors; however, this is an area of ongoing research, and warrants further study as more data become available. Similar approaches will be studied for other animal sub-groups.

Additional data would help confirm and track diet changes over time, which are used to introduce variability in VS production for beef and dairy cows, heifers, and steers. A similar approach for swine volatile solids production may improve the accuracy of future inventory estimates. Uncertainty also exists with the maximum CH_4 producing potential of volatile solids excreted by different animal groups (i.e., B_o). The B_o values used in the CH_4 calculations are published values for U.S. animal waste. However, there are several studies that provide a range of B_o values for certain animals, including dairy and swine. The B_o values chosen for dairy assign separate values for dairy cows and dairy heifers to better represent the feeding regimens of these animal groups. For example, dairy heifers do not receive an abundance of high-energy feed and, consequently, their waste will not produce as much CH_4 as would that from milking cows.

An uncertainty analysis was conducted on the manure management inventory considering the issues described above and based on published data from scientific and statistical literature, the IPCC, and experts in the industry. The results of the uncertainty analysis showed that the manure management CH_4 inventory has a 95 percent confidence interval of -18 percent to 20 percent around the inventory value, and the manure management N_2O inventory has a 95 percent confidence interval of -16 percent to 24 percent around the inventory value.

2.9.2 Unmanaged Waste

Actual N_2O emissions from manure deposited on pasture, range, and paddocks depend on N inputs and other soil characteristics, such as organic carbon availability, O_2 partial pressure, soil moisture content, pH, and soil temperature. The combined interaction of these variables on N_2O flux is complex and highly uncertain. Therefore, the IPCC default methodology, which is used here, is based only on N inputs and does not consider soil characteristics. In addition, N

inputs are estimated from livestock waste excretion rates, which are based on population and weight statistics.

2.10 Mitigating Greenhouse Gas Emissions from Livestock

2.10.1 Enteric Fermentation

Emissions of CH_4 from enteric fermentation in ruminant and non-ruminant animals are dependent on the animal's digestive system and the amount and type of feed consumed. On average, beef and dairy cattle use 6 percent of gross energy intake from feed on enteric fermentation, constituting a loss of energy from the perspective of the animal (Johnson and Johnson 1995). Research on animal nutrition has focused on reducing this energy loss, which consequently reduces CH_4 emissions and increases nutritional efficiency. Through such research, a number of potential strategies have been identified to reduce CH_4 emissions from enteric fermentation, including (Mosier et al. 1998b):

- Increasing the digestibility of forages and feeds;
- Providing feed additives which may tie up hydrogen in the rumen;
- Inhibiting the formation of CH_4 by rumen bacteria;
- Increasing acetic acid in the rumen;
- Improving production efficiency; and
- Modifying bacteria in the rumen.

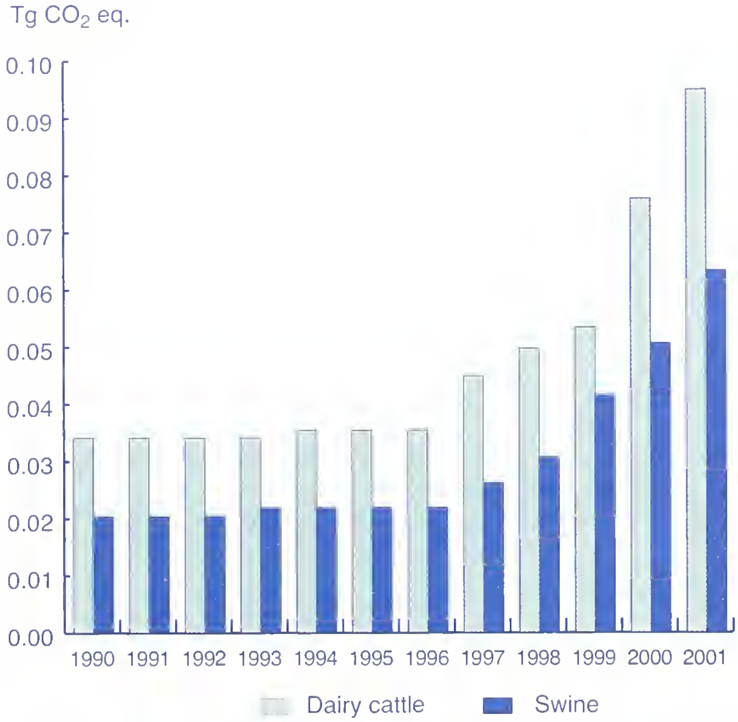
Currently, government research programs indirectly address mitigation of methane emissions through improved livestock production. Ongoing research development and deployment efforts related to mitigating CH_4 emissions include:

- Decreasing feed digestion time by improving grazing management to increase the digestibility of forages, increasing the digestibility of feed grains, and increasing the feeding of concentrated supplements;
- Adding edible oils in feed to sequester hydrogen making it unavailable for methanogens;
- Using feed additives, ionophores, which inhibit the formation of CH_4 by rumen bacteria;
- Improving livestock production efficiency by feed additives such as hormones to increase milk production and growth regulators for beef production or by improved diet or genetics;
- Enhancing rumen microbes to produce usable products rather than CH_4.

2.10.2 Livestock Waste

Livestock and poultry waste from production facilities has the potential to produce significant quantities of CH_4 and N_2O, depending on the waste management practices used. In the United States, livestock and poultry manure is managed in a myriad of ways, suggesting there are multiple options for reducing CH_4 and N_2O emissions. When manure is stored or treated in systems that promote anaerobic conditions, such as lagoons and tanks, the decomposition of the

Figure 2-7
Estimated reductions in methane emissions from anaerobic digesters, 1900-2001



biodegradable fraction of the waste tends to produce CH_4. When manure is handled as a solid, such as in stacks or deposits on pastures, the biodegradable fraction tends to decompose aerobically and produce little or no methane, although it produces N_2O.

A relatively large percent of CH_4 is emitted from livestock and poultry waste in anaerobic lagoons. In 2001, about 14 Tg CO_2 eq. or 35 percent of CH_4 emissions from livestock and poultry waste were from anaerobic lagoons. Current, commercially available technologies that have been the most successful in reducing methane emissions from manure management are anaerobic digestion systems. Unlike conventional lagoons, digestion technologies keep waste treatment and storage functions separate and allow for gas recovery and combustion, pathogen and organic stabilization, odor and other air quality pollution control, and flexible approaches to nutrient management.

EPA tracks installation and usage of anaerobic digesters under voluntary programs, and used these data to estimate how much anaerobic digesters have reduced overall CH_4 emissions from livestock waste for the last 11 years. Figure 2-7 shows an increasing trend in emissions reductions annually from the use of anaerobic digesters, reflecting increasing numbers of digester systems being installed each year. Even so, the reductions achieved to date are less than 1 percent of overall CH_4 emissions from livestock waste.

Other emission reduction processes can include separation, aeration, or shifts to solid handling or storage management systems. These strategies, however, could be limited by other farm or environmental constraints and costs.

Chapter 3: Cropland Agriculture

3.1 Sources of Greenhouse Gas Emissions in Cropland Agriculture

Cropland agriculture results in GHG emissions from multiple sources, with the magnitude of emissions determined, in part, by land management practices. Field burning of crop residues, cultivation of rice, and cultivation and management of soils leads to emissions of N_2O, CH_4, and CO_2. However, agricultural soils can also mitigate GHG emissions through the biological uptake of organic carbon in soils resulting in CO_2 removals from the atmosphere. This chapter covers both GHG emissions from cropland agriculture and biological uptake of CO_2 in agricultural soils. National estimates of these sources, published in the U.S. GHG Inventory, are reported in this section and, where appropriate, State-level emissions estimates are provided.

3.1.1 Residue Burning

Crop residues are sometimes burned in fields to prepare for cultivation and control for pests and disease, although this is not a common practice (EPA 2003a). While CO_2 is a product of residue combustion, residue burning is not considered a net source of CO_2 to the atmosphere. This is because CO_2 released from burning crop biomass is replaced by uptake of CO_2 in crops growing the following season (IPCC 1996). However, CH_4 and N_2O, also products of residue combustion, are not recycled back into crop biomass through biological uptake. Therefore, residue burning is considered a net source of CH_4 and N_2O to the atmosphere. GHG emissions from field burning of crop residues are relatively small in the U.S. (EPA 2003a).

3.1.2 Rice Cultivation

Rice cultivation is unique because it takes place almost universally on flooded fields and in the United States rice is grown exclusively on shallow, continuously flooded fields (EPA 2003a). This water regime causes CH_4 emissions because waterlogged soils create conditions for anaerobic decomposition of organic matter, facilitated by CH_4 emitting "methanogenic" bacteria (IPCC 1996). CH_4 from rice fields reaches the atmosphere in three ways: bubbling up through the soil, diffusion losses from the water surface, and diffusion through the vascular elements of plants (IPCC 1996). Diffusion through plants is considered the primary pathway, with diffusion losses from surface water being the least important process (IPCC 1996). Soil composition, texture and temperature are important variables affecting CH_4 emissions from rice cultivation, as are the availability of carbon substrate and other nutrients, soil pH, and partial pressure of CH_4 (IPCC 1996). Because U.S. rice acreage is relatively small, CH_4 emissions from rice cultivation are small relative to other cropland agriculture sources (EPA 2003a).

3.1.3 Agricultural Soils

Agricultural soils, including cropland and grazing land, serve as both a source of GHGs and a mechanism to remove CO_2 from the atmosphere. Both N_2O emissions and CO_2 emissions and sinks are a function of underlying biological processes. N_2O is produced as an intermediate to natural nitrification and denitrification processes in the soil. In nitrification, soil micro-organisms ("microbes") convert ammonium to nitrate through aerobic oxidation (IPCC 1996). In denitrification, microbes convert nitrate to dinitrogen gas by anaerobic reduction. During

nitrification and denitrification, soil microbes release N_2O, which eventually reaches the atmosphere (IPCC 1996). Cropland soil amendments that add nitrogen to soils drive the production of N_2O by providing additional substrate for nitrification and denitrifcation. Commercial fertilizer, livestock manure, sewage sludge, incorporation of crop residues, and cultivation of nitrogen-fixing crops all add nitrogen to soils. In addition, cultivating highly organic soils (i.e., histosols) enhances mineralization of nitrogen-rich organic matter, making more nitrogen available for nitrification and denitrification (EPA 2003a).

Nitrogen can be converted to N_2O and emitted directly in agricultural fields, or it can be transported through groundwater and runoff to other systems where it is later converted to N_2O, thus causing indirect emissions (IPCC 1996). Some applied nitrogen is volatilized into the atmosphere and subsequently deposited back onto soils, serving as additional, indirect sources of N_2O (EPA 2003a). N_2O from cropland soil amendments is the largest net source of U.S. GHG emissions from cropland agriculture (EPA 2003a).

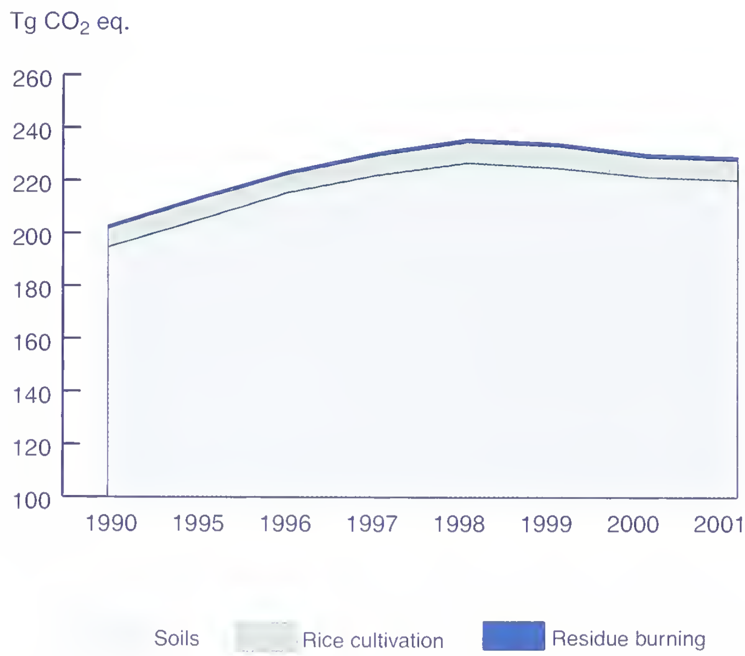
The size of CO_2 emissions and sinks in soils is related to the amount of organic carbon stored in soils (IPCC 1996). Changes in soil organic carbon content are related to inputs, e.g., atmospheric CO_2 fixed as carbon in plants through photosynthesis, and losses mainly driven by decomposition of soil organic matter causing CO_2 emissions (IPCC 1996). The net balance of CO_2 uptake and loss in soils is driven in part by biological processes, which are affected by soil characteristics and climate. In addition, land use and management affect the net balance of CO_2 through modifying inputs and rates of decomposition (IPCC 1996). Changes in agricultural practices such as clearing, drainage, tillage, crop selection, grazing, crop residue management, fertilization, and flooding can modify both organic matter inputs and decomposition, and thereby result in a net flux of CO_2 to or from soils.

Most agricultural soils contain comparatively low amounts of organic carbon as a percentage of total soil mass, typically in the range of 0.5 to 3 percent in the upper 20-30 cm. However, on an area basis this amount of carbon typically exceeds that stored in vegetation in most ecosystems (including forests). Historically, conversion of native ecosystems to agricultural uses resulted in large soil carbon losses, as much as 30-50 percent or more (Haas et al. 1957, Schlesinger 1986). However, after many decades of cultivation, most soils have likely stabilized at lower carbon levels or are increasing their organic matter levels as a result of increasing crop productivity (providing more residues), less intensive tillage and other changes in agricultural management practices (Paustian et al. 1997ab, Allmaras et al. 2000, Follett 2001). Changes in land-use or management practices that result in increased organic inputs or decreased oxidation of organic matter (e.g., taking cropland out of production, improved crop rotations, cover crops, application of organic amendments and manure, and reduction or elimination of tillage) will result in a net accumulation of soil organic carbon until a new equilibrium is achieved.

Cultivation of highly organic soils and additions of carbonate-based lime amendments cause

CO_2 losses to the atmosphere. Cultivated organic soils, also referred to as histosols, contain more than 20 to 30 percent organic matter by weight, and constitute a special case. Organic soils form under water-logged conditions, in which decomposition of plant residues is retarded. When organic soils are drained and cultivated the rate of decomposition and hence CO_2 emissions are greatly accelerated. Because of the depth and richness of the organic layers, carbon loss from cultivated organic soils can continue over long periods of time. Unless restored to undrained, anaerobic conditions, cultivated organic soils remain a net source of CO_2.

Figure 3-1
Trends in greenhouse gas emissions from cropland agriculture, 1990, 1995-2001



In addition, lime, often added to mineral and organic agricultural soils to reduce acidic conditions, contains carbonate compounds (e.g., limestone and dolomite) that when added to soils release CO_2 through the bicarbonate equilibrium reaction (IPCC 1996).

3.1.4 Agroforestry

Agroforestry practices such as establishing windbreaks and riparian forest buffers represent another potential carbon sink in cropland agriculture. Comprehensive data on agroforestry practices are not available to estimate the current national levels of carbon sequestration from such practices. However, published research studies have estimated the potential agroforestry carbon sink in the United States. In temperate systems, agroforestry practices store large amounts of carbon (Kort and Turlock 1999, Schroeder 1994), with the potential ranging from 15 to 198 tons of carbon per hectare (modal value of 34 tons of carbon per hectare) (Dixon 1995). Nair and Nair (2003) estimated that by the year 2025, the potential carbon sequestration of agroforestry in the United States is 90.3 million tons of carbon per year. There is a need to better quantify and track agroforestry practices nationally, particularly to inform USDA programs like the Conservation Reserve Program, Environmental Quality Incentives Program, and Forest Land Enhancement Program, which may provide incentives to land owners to implement agroforestry.

Table 3-1 Summary of GHG emissions from cropland agriculture, 1990, 1995-2001

Source	Gas	1990	1995	1996	1997	1998	1999	2000	2001
					Tg CO_2 eq.				
Residue burning	CH_4	0.70	0.70	0.70	0.80	0.80	0.80	0.80	0.80
Residue burning	N_2O	0.40	0.40	0.40	0.40	0.50	0.40	0.50	0.50
Rice cultivation	CH_4	7.10	7.60	7.00	7.50	7.90	8.30	7.50	7.60
Soils	N_2O	207.93	219.77	229.00	235.93	238.16	236.65	235.14	235.40
Soils[1]	CO_2	(13.30)	(14.90)	(13.60)	(13.90)	(11.50)	(11.90)	(13.80)	(15.20)
Mineral soils		(57.1)	(58.6)	(57.3)	(57.4)	(55.8)	(55.7)	(57.3)	(59.1)
Organic soils		34.3	34.8	34.8	34.8	34.8	34.8	34.8	34.8
Liming of soils		9.5	8.9	8.9	8.7	9.6	9.1	8.8	9.1
Total emissions		259.93	272.17	280.80	288.13	291.76	290.05	287.54	288.20
Net emissions (sources and sinks)		202.83	213.57	223.50	230.73	235.86	234.25	230.14	229.10

Note: Parentheses indicate net sequestration.
[1] Soil carbon sequestration on land under the Conservation Reserve Program and on range and grazing lands is included in the total for mineral soils.

3.2 Summary of U.S. Greenhouse Gas Emissions from Cropland Agriculture

In 2001, cropland agriculture resulted in total emissions of 288 Tg CO_2 eq. of GHG (Table 3-1). GHG emission from agricultural soils, including N_2O and CO_2, were responsible for the majority of total emissions, while residue burning and rice cultivation caused less than 4 percent of emissions (Table 3-1). N_2O and CO_2 from agricultural soils totaled 235 Tg CO_2 eq. and 44 Tg CO_2 eq., respectively, in 2001. However, that amount was offset by the uptake of 59 Tg CO_2 eq. in agricultural soils in 2001. Thus, net emissions of GHGs from cropland agriculture were just under 230 Tg CO_2 eq. Since 1990, the magnitude of residue burning and rice cultivation emissions has remained relatively stable (Figure 3-1). Comparatively, GHG emissions from agricultural soils increased from 1990 to 1998 then decreased each year until 2001 (Table 3-1). Overall, net GHG emissions from cropland agriculture increased 12 percent from 1990 and 2001.

3.3 Residue Burning

GHG emissions from field burning of crop residues are a function of the amount and type of residues burned. In the United States, crops burned include wheat, rice, sugarcane, corn, barley, soybeans, and peanuts (EPA 2003a). For most crops, less than 5 percent of residues are burned

Table 3-2 Greenhouse gas emissions from agriculture burning, by crop type, 1990-2001

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
						Tg CO_2 eq.						
CH_4	0.68	0.64	0.75	0.60	0.81	0.66	0.75	0.76	0.78	0.76	0.78	0.76
Wheat	0.14	0.10	0.12	0.12	0.12	0.11	0.11	0.12	0.13	0.12	0.11	0.10
Rice	0.08	0.08	0.08	0.08	0.10	0.08	0.09	0.07	0.06	0.07	0.07	0.07
Sugarcane	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02
Corn	0.28	0.27	0.34	0.23	0.36	0.26	0.33	0.33	0.35	0.34	0.35	0.34
Barley	0.02	0.02	0.02	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
Soybeans	0.15	0.15	0.17	0.14	0.19	0.17	0.18	0.21	0.21	0.20	0.21	0.22
Peanuts	-	-	-	-	-	-	-	-	-	-	-	-
N_2O	0.37	0.36	0.41	0.34	0.45	0.38	0.42	0.45	0.45	0.44	0.46	0.46
Wheat	0.05	0.03	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.03
Rice	0.04	0.04	0.04	0.04	0.05	0.04	0.04	0.03	0.03	0.03	0.03	0.03
Sugarcane	-	-	-	-	-	-	-	-	0.01	0.01	0.01	0.01
Corn	0.09	0.08	0.11	0.07	0.11	0.08	0.10	0.10	0.11	0.11	0.11	0.11
Barley	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	-	-	-
Soybeans	0.18	0.19	0.21	0.18	0.24	0.21	0.23	0.26	0.26	0.25	0.26	0.28
Peanuts	-	-	-	-	-	-	-	-	-	-	-	-
Total	1.05	1.00	1.16	0.94	1.26	1.03	1.17	1.21	1.24	1.20	1.24	1.22

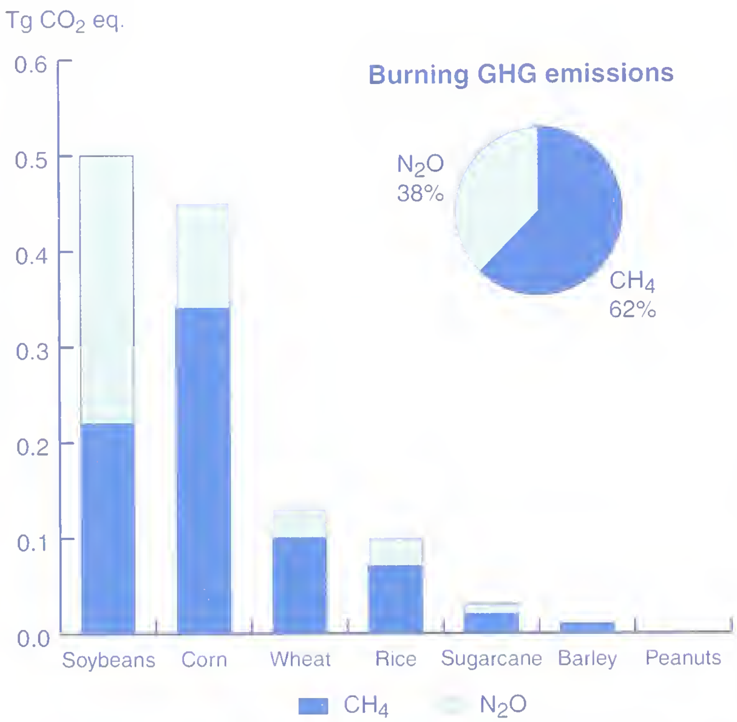
- = less than 0.005

per year; a higher portion of rice residues is burned annually (EPA 2003a).

About two-thirds of GHG emissions from residue burning were CH_4 across all crop types, in 2001; the remaining third were N_2O (Table 3-2; Figure 3-2). The highest GHG emissions were from burning of soybean and corn crop residues, at 40 percent and 30 percent respectively. Burning of wheat, rice, sugarcane, and barely crop residues each contributed 10 percent or less to overall GHG emissions; burning of peanut crop residues contributed almost nothing to this source of GHGs.

Total GHG emissions from residue burning increased 17 percent from 1990 to 2001. Trends in relative GHG emissions were similar across crop types in 1990 compared to 2001 with a few exceptions. In 1990, burning of corn residues contributed the most to GHG emissions from residue burning, while burning of soybeans was the second largest source (Figure 3-3). By

Figure 3-2
Greenhouse gas emissions from burning by crop type, 2001



2001, these positions reversed, corresponding to changes in production. Between 1990 and 2001, soybean and corn production both increased in absolute amounts [Figure 3-4(A)]. However, proportionally, soybean production increased more dramatically than corn (soybean production increased by 50 percent and corn by 20 percent) [Figure 3-4(B)]. In addition, soybeans have higher nitrogen content than corn, resulting in greater N_2O emission per unit of crop mass burned (Appendix Table B-1). Thus, while corn production was still greater than soybean production in 2001, GHG emissions from soybean residue burning exceeded those from corn residue burning. Appendix Table B-2 provides the complete time series of crop production from 1990 to 2001 for crop types that contribute to GHG emissions from burning.

Illinois and Iowa had the highest State levels of GHG emissions from residue burning in 2001, with each emitting roughly 0.16 Tg CO_2 eq. of CH_4 and N_2O combined (Appendix Table B-4 and Appendix Table B-5). The next highest levels of GHG emissions from residue burning were in Nebraska, Indiana, Minnesota, Arkansas, Ohio, Kansas, Missouri, and South Dakota, with emissions between 0.04 and 0.1 Tg CO_2 eq. State-level GHG emissions from residue burning are strongly tied to crop production. State-level estimates of crop production are provided in Appendix Table B-3 for corn, soybeans, wheat, rice, sugarcane, barley, and peanuts.

3.4 Methods for Estimating CH_4 and N_2O Emissions from Residue Burning

EPA provided national-level estimates of GHG emissions from agricultural residue burning for all crop types except rice, and State-level estimates for GHG emissions from rice residue burning for this report. In addition, State-level estimates were derived by USDA for all crop types (except rice) using the same method. Details on the methods used by EPA are provided below, including excerpts from Chapter 5 of the U.S. GHG Inventory report (EPA 2003a). This

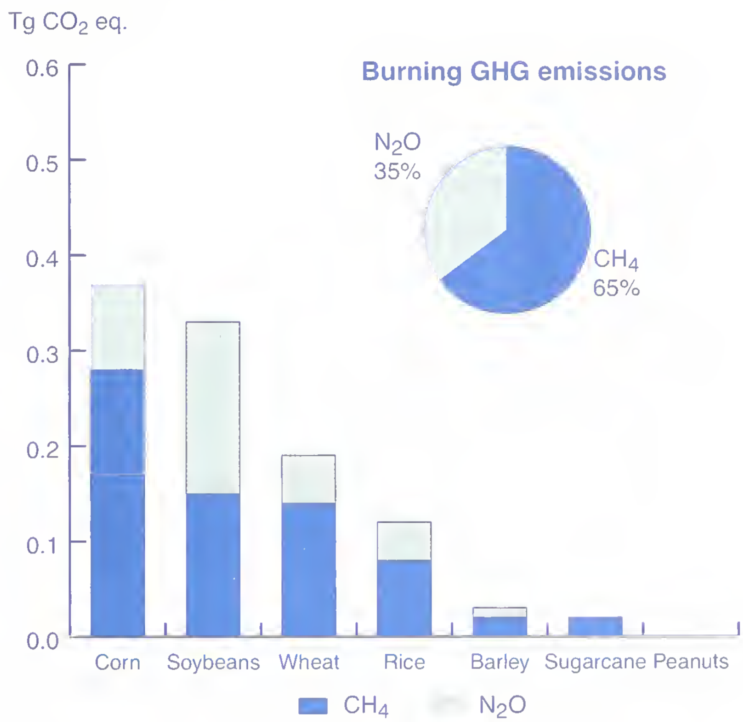
information is reproduced with permission from EPA.

The equations below were used to estimate the amounts of carbon and nitrogen released during burning. Final emissions estimates were derived from the amount of carbon and nitrogen released using emissions ratios for CH_4, CO, N_2O, and NO_x published by the IPCC.

Carbon Released = (Annual Crop Production) × (Residue/Crop Product Ratio) × (Fraction of Residues Burned in situ) × (Dry Matter Content of the Residue) × (Burning Efficiency) × (Carbon Content of the Residue) × (Combustion Efficiency)

Nitrogen Released = (Annual Crop Production) × (Residue/Crop Product Ratio) × (Fraction of Residues Burned in situ) × (Dry Matter Content of the Residue) × (Burning Efficiency) × (Nitrogen Content of the Residue) × (Combustion Efficiency)

Figure 3-3
Greenhouse gas emissions from burning by crop type, 1990



Values used in the above equations to estimate emissions from residue burning are summarized in Appendix Table B-1. National and State-level crop production statistics are provided in Appendix Table B-2 and Appendix Table B-3. The sources for developing these input data are described for each parameter below.

Annual Crop Production: The crop residues that are burned in the United States were determined from various State-level GHG emission inventories (ILENR 1993, Oregon Department of Energy 1995, Wisconsin Department of Natural Resources 1993) and publications on agricultural burning in the United States (Jenkins et al. 1992, Turn et al. 1997, EPA 1992b). Crop production data for these crops, except rice in Florida, were taken from USDA NASS (Field Crops: 1994, 1998; Crop Production 2001, 2000). Rice production data for Florida were estimated by applying average primary and ratoon crop yields for Florida (Smith 1999) to Florida acreages (Schueneman 1999b, 2001; Deren 2002).

Figure 3-4(A)
Change in commodity production, 1990-2001

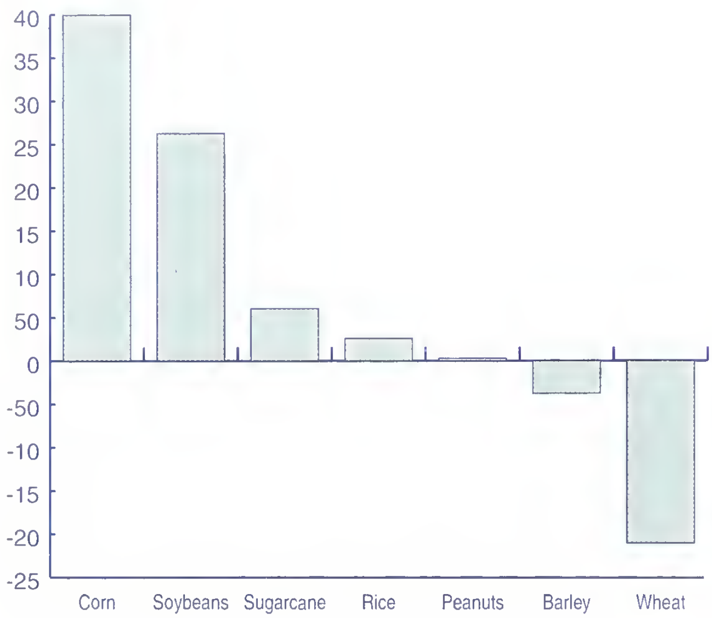
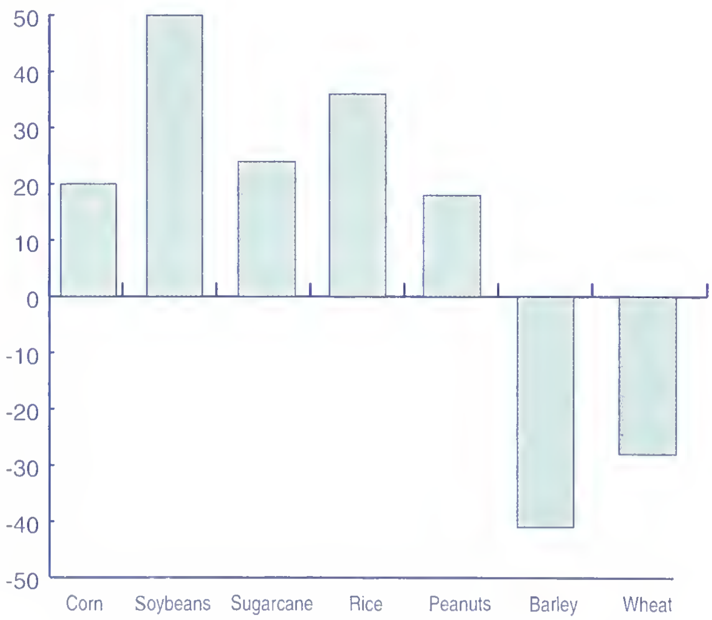


Figure 3-4(B)
Percent change in commodity production, 1990-2001



Residue-to-Crop Product Mass Ratios: All residue/crop product mass ratios except sugarcane were obtained from Strehler and Stützle (1987). The ratio for sugarcane is from the University of California (1977).

Fraction of Residues Burned: The percentage of crop residue burned was assumed to be 3 percent for all crops in all years, except rice, based on State inventory data (ILENR 1993, Oregon Department of Energy 1995, Noller 1996, Wisconsin Department of Natural Resources 1993, and Cibrowski 1996). Estimates of the percentage of rice acreage on which residue burning took place were obtained on a State-by-State basis from agricultural extension agents in each of the seven rice-producing States (Bollich 2000; Deren 2002; Guethle 1999, 2000, 2001, 2002; Fife 1999; California Air Resources Board 1999; Klosterboer 1999a, 1999b, 2000, 2001, 2002; Linscombe 1999a, 1999b, 2001, 2002; Mutters 2002, Najita 2000, 2001; Schueneman 1999a, 1999b, 2001; Slaton 1999a, 1999b, 2000; Street 1999a, 1999b, 2000, 2001, 2002; Wilson 2001, 2002) (Appendix Table B-1).

The estimates provided for Arkansas and Florida remained constant over the entire 1990-2001 period, while the estimates for all other States varied over the time series. For California, it was assumed that the annual percent of rice acreage burned in Sacramento Valley is

representative of burning in the entire State, because the Sacramento Valley accounts for over 95 percent of the rice acreage in California (Fife 1999). The annual percent of rice acreage burned in the Sacramento Valley was obtained from staff at the California Air Resources Board (CARB) (Najita, 2001), a report of the CARB (2001), and background data for future editions of the report (Lindberg 2002). These values declined over the period 1990 through 2001 because of a legislated reduction in rice straw burning.

Residue Dry-Matter Content: Residue dry-matter contents for all crops except soybeans and peanuts were obtained from Turn et al. (1997). Soybean dry-matter content was obtained from Strehler and Stützle (1987). Peanut dry-matter content was obtained through personal communications with Jen Ketzis (1999), who accessed Cornell University's Department of Animal Science's computer model, Cornell Net Carbohydrate and Protein System.

Burning Efficiency: Burning efficiency refers to the fraction of dry biomass exposed to burning that actually burns. The burning efficiency was assumed to be 93 percent.

Carbon and Nitrogen Content: The residue carbon contents and nitrogen contents for all crops except soybeans and peanuts are from Turn et al. (1997). The residue carbon content for soybeans and peanuts is the IPCC default (IPCC/UNEP/OECD/IEA 1997). The nitrogen content of soybeans is from Barnard and Kristoferson (1985). The nitrogen content of peanuts is from Ketzis (1999).

Combustion Efficiency: Combustion efficiency refers to the fraction of carbon in the fire that is oxidized completely to CO_2. Combustion efficiency was assumed to be 88 percent for all crop types (EPA 1994).

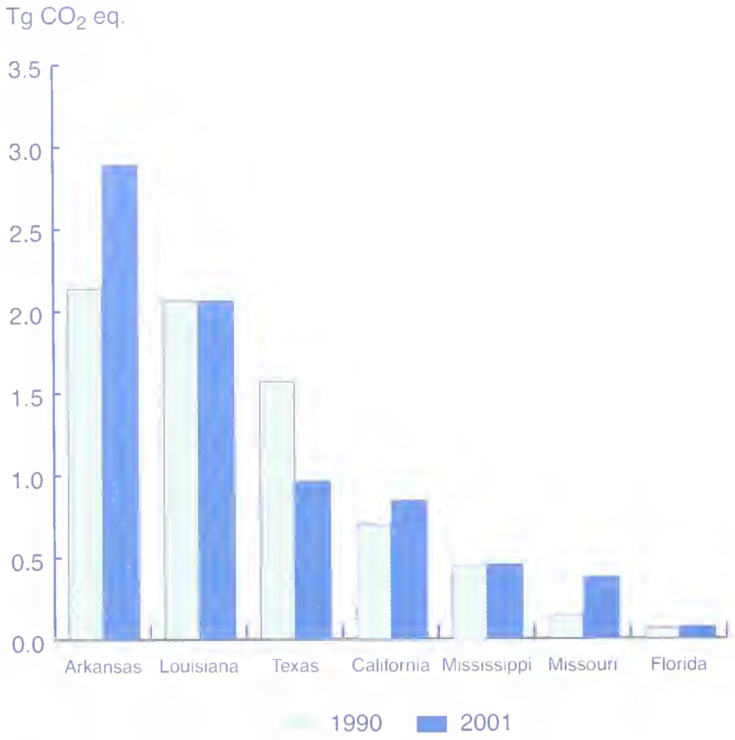
State-level emissions estimates were calculated with the above equations, applying State-level production data to national-level coefficients. The State-level rice estimates were provided directly by EPA, using State-specific residue fractions for rice because the fraction of residues burned varies among States for rice, and State production data.

3.5 Uncertainty in Estimating CH_4 and N_2O Emissions from Residue Burning

The following discussion of uncertainty in estimating GHG emissions from residue burning is modified from information provided in the U.S. GHG Inventory. The information is reproduced here with permission from EPA.

Assumptions about the annual amount of residues burned by crop type are the largest source of uncertainty in estimating GHG emissions from field burning of agricultural residues. Data on the fraction burned, as well as the gross amount of residue burned each year, are not collected at either the national or State level. In addition, burning practices are highly variable among crops and States. The fractions of residue burned used in these calculations were based upon information collected by State agencies and in the published literature. These emissions

Figure 3-5
Methane from rice cultivation by State, 1990 and 2001



estimates may continue to change as more information becomes available in the future. Other sources of uncertainty include the residue/crop product mass ratios, residue dry matter contents, burning and combustion efficiencies, and emission ratios. Residue/crop product ratios for specific crops can vary among cultivars and, for all crops except sugarcane, generic global residue/crop product ratios were used rather than ratios specific to the United States. In addition, residue dry matter contents, burning and combustion efficiencies, and emission ratios can vary due to weather and other combustion conditions, such as fuel geometry. Values for these variables were taken from literature on agricultural biomass burning.

3.6 Rice Cultivation

CH_4 emissions from rice cultivation[10] are limited to seven U.S. States. In four States, the climate allows for cultivation of two rice crops per season, the second of which is referred to as a ratoon crop (EPA 2003a). CH_4 emissions from primary and ratoon crops are accounted for separately because emissions are higher from ratoon crops (EPA 2003a). Overall, rice cultivation is a small source of CH_4 in the United States. In 2001, CH_4 emissions totaled 7.6 Tg CO_2 eq, of which 5.9 Tg CO_2 eq. were from primary crops in all seven States and 1.7 Tg CO_2 were from ratoon crops in four States (Appendix Table B-7).

Arkansas and Louisiana had the highest CH_4 emissions from rice cultivation in 2001, followed by Texas and California. Mississippi, Missouri, and Florida each had emissions of less than 0.5 Tg CO_2 eq. (Figure 3-5). Since 1990, CH_4 emissions from rice cultivation have increased 7 percent. While small national changes were seen between 1990 and 2001, sizeable shifts occurred at State levels during that time period. For example, CH_4 emission in Arkansas and California increased by 35 percent and 19 percent, respectively, while emissions in Texas declined by 39 percent (Figure 3-5 and Table 3-3). CH_4 emissions from Missouri increased by

[10] This source focuses on CH_4 emissions resulting from anaerobic decomposition, and does not include emissions from burning of rice residues. The later is covered in section 3.3.

over 150 percent between 1990 and 2001, but remained small in magnitude relative to emissions from other States. State-level shifts in CH_4 emissions since 1990 are positively correlated with changes in area of rice cultivation (Appendix Table B-6). Appendix Table B-6 provides a complete time series of areas harvested for rice by State and primary versus ratoon crops from 1990-2001.

Table 3-3 Change in methane emissions from rice cultivation, 1990-2001

	1990	2001	Change, 1990-2001
	Tg CO_2 eq.		*percent*
Arkansas	2.14	2.89	35
California	0.70	0.84	19
Florida	0.06	0.07	4
Louisiana	2.06	2.06	0
Mississippi	0.45	0.45	1
Missouri	0.14	0.37	159
Texas	1.57	0.96	-39
Total	7.12	7.64	7

3.7 Methods for Estimating CH_4 Emissions from Rice Cultivation

EPA provided estimates for CH_4 emissions from rice cultivation for this report. Details on the methods are provided below and are excerpted, with permission from EPA, from Chapter 5 of the U.S. GHG Inventory report (EPA 2003a). The method used by EPA applies area-based seasonally integrated emission factors (i.e., amount of CH_4 emitted over a growing season per unit harvested area) to harvested rice areas to estimate annual CH_4 emissions from rice cultivation. EPA derived specific CH_4 emission factors from published studies containing rice field measurements in the United States, with separate emissions factors for ratoon and primary crops to account for higher seasonal emissions in ratoon crops.

A review of published experiments was used to develop emissions factors for primary and ratoon crops. Experiments where nitrate or sulfate fertilizers or other substances believed to suppress CH_4 formation were applied, and experiments where measurements were not made over an entire flooding season or where floodwaters were drained mid-season, were excluded from the analysis. The remaining experimental results were then sorted by season (i.e., primary and ratoon) and type of fertilizer amendment (i.e., no fertilizer added, organic fertilizer added, and synthetic and organic fertilizer added). The experimental results from primary crops with synthetic and organic fertilizer added (Bossio et al. 1999, Cicerone et al. 1992, Sass et al. 1991a and 1991b) were averaged to derive an emission factor for the primary crop, and the experimental results from ratoon crops with synthetic fertilizer added (Lindau and Bollich 1993, Lindau et al. 1995) were averaged to derive an emission factor for the ratoon crop. The resultant emission factor for the primary crop is 210 kg CH4/hectare-season, and the resultant emission factor for the ratoon crop is 780 kg CH_4/hectare-season.

The harvested rice areas for the primary and ratoon crops in each State are presented in Appendix Table B-6. Primary crop areas for 1990 through 2001 for all States except Florida

were taken from USDA NASS Field Crops Final Estimates 1987-1992 (USDA NASS Field Crops 1994), Field Crops Final Estimates 1992-1997 (USDA NASS Field Crops 1998), Crop Production 2000 Summary (USDA NASS Crop Production 2001), and Crop Production 2001 Summary (USDA NASS Crop Production 2002). Harvested rice areas in Florida, which are not reported by USDA, were obtained from Tom Schueneman (1999b, 1999c, 2000, 2001a), a Florida agricultural extension agent, and Dr. Chris Deren (2002) of the Everglades Research and Education Center at the University of Florida. Acreages for the ratoon crops were derived from conversations with the agricultural extension agents in each State.

In Arkansas, ratooning occurred only in 1998 and 1999, when the ratoon area was less than 1 percent of the primary area (Slaton 1999, 2000, 2001a). In Florida, the ratoon area was 50 percent of the primary area from 1990 to 1998 (Schueneman 1999a), about 65 percent of the primary area in 1999 (Schueneman 2000), around 41 percent of the primary area in 2000 (Schueneman 2001a), and about 70 percent of the primary area in 2001(Deren 2002). In Louisiana, the percentage of the primary area in ratoon was constant at 30 percent over the 1990 to 1999 period, but increased to approximately 40 percent in 2000, before returning to 30 percent in 2001 (Linscombe 1999a, 2001a, 2002 and Bollich 2000). In Texas, the percentage of the primary area in ratoon was constant at 40 percent over the entire 1990 to 1999 period and in 2001, but increased to 50 percent in 2000 due to an early primary crop (Klosterboer 1999, 2000, 2001a, 2002).

3.8 Uncertainty in Estimating CH_4 Emissions from Rice Cultivation

The following discussion of uncertainty in estimating GHG emissions from rice cultivation is modified from information provided in the U.S. GHG Inventory. The information is reproduced here with permissions from EPA.

CH_4 emissions factors are the largest source of uncertainty in estimates for rice cultivation. Seasonal emissions, derived from field measurements in the United States, vary by more than an order of magnitude, from variation in cultivation practices, fertilizer application, cultivar types, and soil and climatic conditions. Some variability is accounted for by separating primary from ratoon areas. However, even within a cropping season, measured emissions vary significantly. Of the experiments that were used to derive the emission factors used here, primary emissions ranged from 22 to 479 kg CH_4/hectare-season and ratoon emissions ranged from 481 to 1,490 kg CH_4/hectare-season. Based on these emission ranges, total CH_4 emissions from rice cultivation in 2001 were estimated to range from 1.7 to 17 Tg CO2 eq.

In addition, data are not collected regularly on the area of rice crops in ratoon, creating another relatively minor source of uncertainty. The area estimates are derived from expert opinion and account for less than 10 percent of the total area of rice cultivation. A final source of uncertainty is the practice of flooding outside of the normal rice season. According to agriculture extension agents, this occurs in all rice-growing States. Estimates of the area of off-season flooding range from 5 to 68 percent of the rice acreage. Fields are flooded for a variety

of reasons: to provide habitat for waterfowl, to provide ponds for crawfish production, and to aid in rice straw decomposition.

3.9 Agricultural Soils

Sources of N_2O are addressed in this section, including soil amendments (commercial fertilizer, livestock manure, crop residues, and sewage sludge), nitrogen-fixing crops, and histosol cultivation. National-level emissions estimates are discussed; State-level data for these sources are not yet available, although work is underway to develop process models that can make finer-scale estimates of N_2O and other GHG emissions from agriculture (Box 3-1). CO_2 emissions and sinks in agricultural soils are discussed in the following section.

3.9.1 Commercial Fertilizer

Commercial fertilizers were the largest source of N_2O emissions in cropland soils in 2001, emitting over 100 Tg CO_2 eq. of N_2O (Figure 3-6, Table 3-4). The majority of emissions from commercial fertilizers were direct, while some were from leaching, and a small portion was from volatilization. The methods used to estimate emissions from fertilizer assume that all fertilizer purchased in the United States is used in cropland agriculture, which is a simplifying assumption necessitated by limited data. Thus, the numbers reported here and in the U.S. GHG Inventory could overestimate emissions attributable to cropland soils. Total N_2O emissions from fertilizer are more likely apportioned across a range of land uses where fertilizer is applied. For example, fertilizer is applied to forests and lawns—some estimates suggest as much as 6 percent of U.S. fertilizer consumption is for turfgrass (EPA 1999).

Box 3-1

National Inventory Development and Assessment of Soil-Atmosphere Exchange of N_2O, NO_x and CH_4 Oxidation in U.S. Agricultural Soils

A.R. Mosier (USDA – ARS), W.J. Parton (Colo. St. Univ.), S. J. Del Grosso (Colo. St. Univ.) and T. Wirth (EPA)

A collaborative agreement between EPA and USDA/ARS and the Colorado State University Natural Resource Ecology Laboratory is laying the groundwork for using the ecosystem-process-based model DAYCENT to conduct the U.S. national inventory for N_2O emissions from agricultural soils. Over the next few years the model-based inventory will be developed and compared and contrasted to the IPCC Agricultural Soils N_2O estimation that is now used in the national inventory.

DAYCENT Model Description

The DAYCENT ecosystem model (Parton et al., 1998; Kelly et al., 2000; Del Grosso et al., 2001) simulates exchanges of carbon (C), nutrients (N, P, S), and gases (CO_2, CH_4, N_2O, NO_x, N_2) among the atmosphere, soil, and plants. Required inputs to drive the model include daily maximum/minimum temperature and precipitation, site-specific soil properties, and current and historical land use. Disturbances and management practices such as fire, grazing, cultivation, and organic matter or fertilizer additions can be simulated. The submodels used in DAYCENT are described in detail by Del Grosso et al. (2001) and the model code is available from the authors.

DAYCENT includes submodels for plant productivity, decomposition of dead plant material and soil organic matter (SOM), soil water and temperature dynamics, and trace gas fluxes. Flows of C and nutrients are controlled by the amount of C in the various pools (e.g. SOM, plant biomass), the N and lignin concentrations of the pools, abiotic temperature/soil water factors, and soil physical properties related to texture. SOM is divided into three pools based on decomposition rates (Parton et al., 1993; 1994). Decomposition of SOM and external nutrient additions supply the nutrient pool available for plant growth and microbial processes resulting in trace gas fluxes. Plant growth is controlled

(Continued on page 58)

(Continued from page 57)
by: a plant-specific maximum growth parameter, nutrient availability, and 0-1 multipliers that reflect shading, water, and temperature stress. Net Primary Productivity (NPP) is allocated among leafy, woody, and root compartments as a function of plant type, season, soil water content, and nutrient availability (Metherell et al., 1993). The land surface submodel of DAYCENT simulates water flow and evapotranspiration for the plant canopy, litter, and soil profile, as well as soil temperature throughout the profile (Parton et al., 1998; Eitzinger et al., 2000).

The trace gas submodel of DAYCENT simulates N_2O, NO_x, and N_2 emissions from soils resulting from nitrification and denitrification as well as CH_4 oxidation in soils. The nitrification submodel simulates N_2O and NO_x emissions as a function of soil NH_4., water content, temperature, pH, and texture (Parton et al., 2001). Nitrification is limited by moisture stress when soil water-filled pore space (WFPS) is too low and by O_2 availability when WFPS is too high. Optimum WFPS for nitrification is ~55%, with a higher optimum for clay than sandy soils.

The denitrification submodel simulates N_2O, N_2, and NO_x emissions as a function of soil NO_3., water content, labile C availability (most denitrifiers are heterotrophs), and soil physical properties related to texture that influence gas diffusion rates (Del Grosso et al., 2000b). Denitrification, an anaerobic process, does not occur until WFPS exceeds 50-60 percent then it increases exponentially as WFPS increases and levels off as soils approach saturation. Simulated heterotrophic respiration rates are used as a proxy for labile C availability.

NO_x emissions are calculated using total N_2O emissions, a NO_x:N_2O function based on soil gas diffusivity, and a pulse multiplier based on rainfall frequency and amount (Parton et al., 2001). As soil gas diffusivity decreases, a smaller proportion of total N gas fluxes are assumed to be in the form of NO_x because NO_x becomes more reactive as soils become more reducing. The pulse multiplier equations were developed by Yienger and Levy (1995) and account for the observed high NO_x emission rates following precipitation events onto soils that were previously dry (Smart et al., 1999; Martin et al., 1998; Hutchinson et al., 1993). CH_4 uptake is controlled by soil gas diffusivity, water content, and temperature (Del Grosso et al., 2000a). CH_4 oxidation rates are assumed to be limited by gas diffusivity when volumetric soil water content is too high and by moisture stress on biological activity when volumetric soil water content is too low. Optimum volumetric soil water content values range from 0.06-0.22 cm^3 H_2O per cm^3 soil. As with nitrification, clay soils are assumed to have higher optimum water content for CH_4 oxidation than sandy soils.

In 2001, total commercial synthetic fertilizer consumption was 10,684 Gg[11] of nitrogen, an increase of 580 Gg of nitrogen (or 6 percent) since 1990. The highest levels of commercial synthetic fertilizer consumption, and therefore emissions, were in 1999 at 11,237 Gg of nitrogen [Figure 3-7 (A)]. Since then, consumption has decreased steadily. Even so, nitrogen applications of commercial synthetic fertilizers were still far greater than those from other sources in cropland soils in 2001.

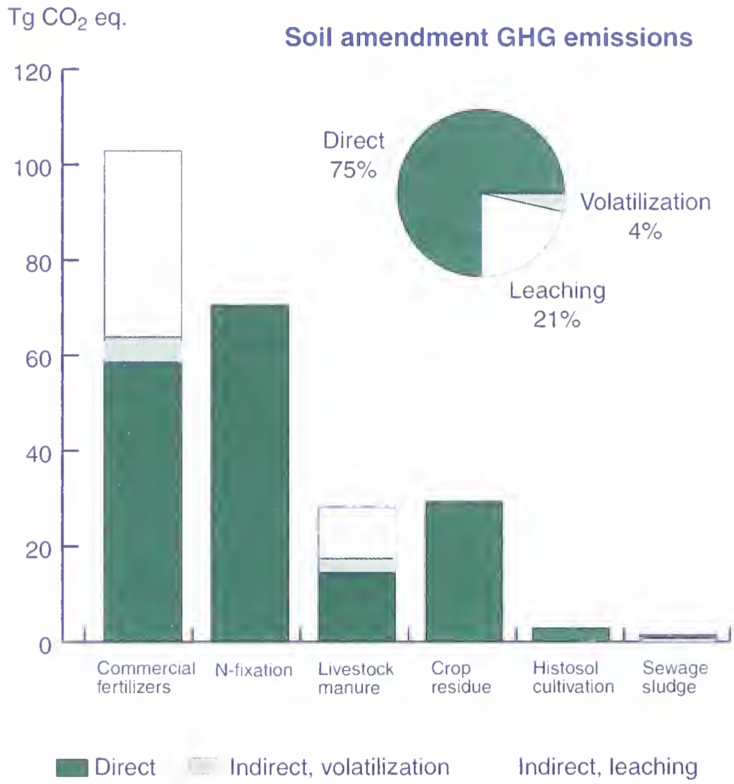
3.9.2 Livestock Manure

Livestock manure application was the fourth largest source of N_2O emissions from cropland soils in 2001 (Figure 3-6, Table 3-4). About half of the 28 Tg CO_2 eq. of N_2O from livestock waste was emitted directly (14 Tg CO_2 eq.). The remaining portion was predominantly indirect from leaching and runoff (11 Tg CO_2 eq.) and a small portion was from volatilization (3 Tg

[11] By convention, activity data are expressed in Gg given their magnitude. One Gg equals 10^{-3} Tg. Box 1-1 gives additional information on units.

CO_2 eq.). N_2O from livestock waste applied to croplands increased 11 percent between 1990 and 2001, with steady increases most years (Table 3-4) tracking the trend in livestock waste application to crops [Figure 3-7 (B)]. While livestock manure is a source of N_2O when applied to soils, it also enhances soil carbon sequestration by providing an input of organic matter to soils. Carbon gains from livestock waste applications essentially offset a portion of GHG emissions. The size of this offset is addressed in the next section on CO_2 emissions and sinks in agricultural soils.

Figure 3-6
Nitrous oxide emissions from agricultural soil management by source and process, 2001



3.9.3 Nitrogen Fixation

Nitrogen-fixing plants were the second largest source of N_2O emissions in cropland soils, contributing about 70 Tg CO_2 eq. in 2001 (Figure 3-6). Nitrogen additions, which are related to nitrogen content in aboveground plant biomass, vary by crop type and production levels (EPA 2003a). While numerous nitrogen fixers are cultivated, emissions from this source are largely attributable to three species of plant: soybeans, white clover, and alfalfa. In 2001, total aboveground biomass nitrogen was highest in soybeans, second highest in white clover, and third highest in alfalfa. Combined, these three crops contributed 93 percent (10,745 Gg) of all nitrogen from nitrogen-fixing crops to soils (Appendix Table B-9). Soybeans and alfalfa are grown in croplands, while white clover is typically grown on pastures for forage. Additional nitrogen fixers include peanuts, dry edible beans, edible peas, lentils, wrinkled seed peas, and Australian winter peas. Red clover, birdsfeet trefoil, arrowleaf clover, and crimson clover are nitrogen fixers grown as forage on pastures.

3.9.4 Crop Residues

The incorporation of crop residues into soils led to N_2O emissions of 29 Tg CO_2 eq. in 2001.

Table 3-4 Nitrous oxide emissions from agricultural soil amendments, 1990-2001

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
						Tg CO_2 eq.						
Commercial fertilizers	97.26	98.93	99.66	103.18	107.45	103.99	107.46	107.63	107.81	108.22	104.94	102.88
Direct emissions	55.40	56.35	56.76	58.77	61.20	59.23	61.20	61.30	61.40	61.64	59.77	58.60
Volatilization emissions	4.93	5.01	5.05	5.23	5.44	5.27	5.45	5.46	5.47	5.49	5.32	5.21
Leaching and run-off emissions	36.93	37.57	37.84	39.18	40.81	39.49	40.81	40.87	40.94	41.10	39.85	39.07
Livestock manure	25.44	26.12	26.01	26.55	26.74	26.46	26.76	27.35	27.68	27.65	28.14	28.27
Direct emissions	13.03	13.38	13.32	13.60	13.69	13.55	13.71	14.01	14.17	14.16	14.41	14.48
Volatilization emissions	2.64	2.71	2.70	2.75	2.77	2.74	2.78	2.84	2.87	2.87	2.92	2.93
Leaching and run-off emissions	9.77	10.03	9.99	10.20	10.27	10.17	10.28	10.50	10.63	10.62	10.81	10.86
Sewage sludge	0.74	0.83	0.93	1.04	1.13	1.23	1.26	1.28	1.30	1.35	1.40	1.40
Direct emissions	0.38	0.43	0.48	0.53	0.58	0.63	0.65	0.66	0.67	0.69	0.72	0.72
Volatilization emissions	0.08	0.09	0.10	0.11	0.12	0.13	0.13	0.13	0.13	0.14	0.14	0.14
Leaching and run-off emissions	0.28	0.32	0.36	0.40	0.44	0.47	0.48	0.49	0.50	0.52	0.54	0.54
N fixation (direct emissions)	58.46	59.52	61.39	57.08	66.07	61.84	63.86	68.15	69.22	68.25	68.80	70.62
Crop residue (direct emissions)	23.22	22.46	26.34	21.11	28.11	23.41	26.80	28.67	29.29	28.31	28.97	29.33
Histosol cultivation (direct emissions)	2.81	2.81	2.80	2.82	2.83	2.84	2.85	2.86	2.87	2.88	2.89	2.90

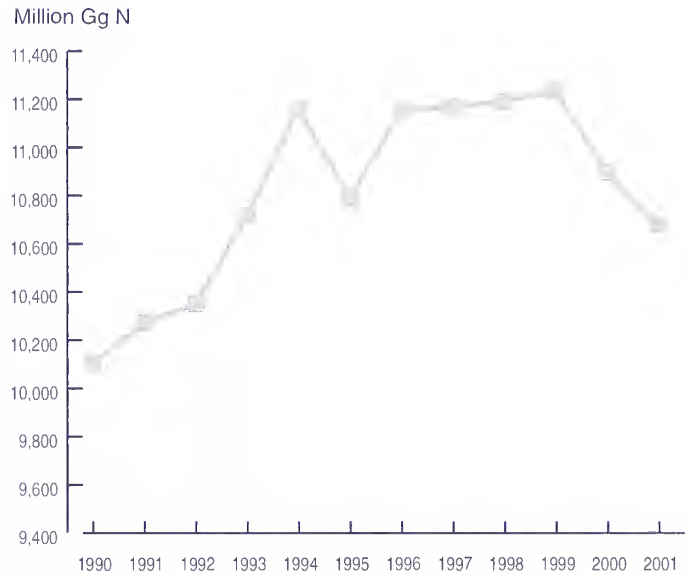
Soybean residues were by far the largest source of nitrogen, adding 2,975 Gg N to soils in 2001 (Appendix Table B-10). Corn was the next largest source at 1,147 Gg N, or about a third that of soybeans. Wheat, sorghum, rice, barley, peanuts, and oats each added between 10 and 400 Gg N, while dry edible peas, rye, lentils, winkled seed peas, and Australian winter peas each contributed less than 10 Gg N.

Soybeans contribute to N_2O emissions both from nitrogen fixation and incorporation of crop residues in the soil. The total contribution of soybeans to GHG emissions increased between 1990 and 2001 as evidenced by the 50 percent increase in nitrogen inputs in both aboveground biomass and crop residues from soybeans (Appendix Table B-9 and Appendix Table B-10). The increase is directly related to increased production of soybeans nationwide. In 2001 soybean production was over 78 million metric tons, up from 52 million metric tons in 1990 (Appendix Table B-2).

3.9.5 Histosol Cultivation

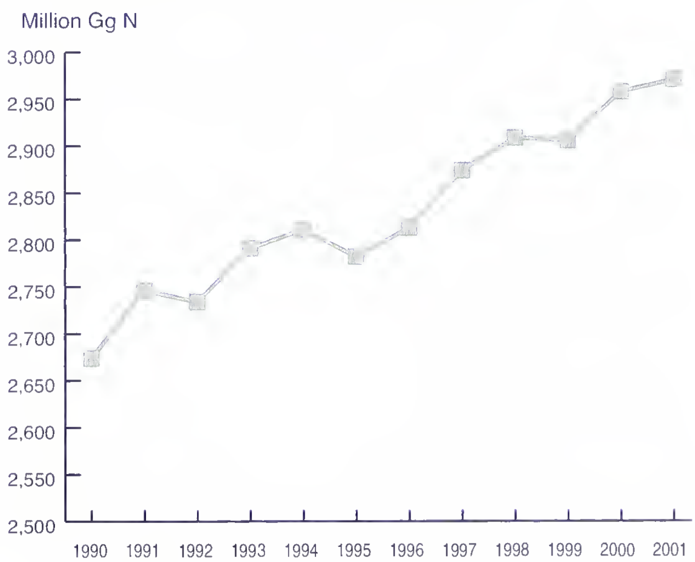
Histosol cultivation is a relatively minor source of N_2O emissions from cropland soils. In 2001, N_2O emissions from this source were 2.9 Tg CO_2 eq, up from 2.8 Tg CO_2 eq. in 1990 (Table 3-4). Sewage sludge is another minor source of N_2O emissions from cropland soils, with

Figure 3-7(A)
Commercial synthetic fertilizer consumption, 1990-2001



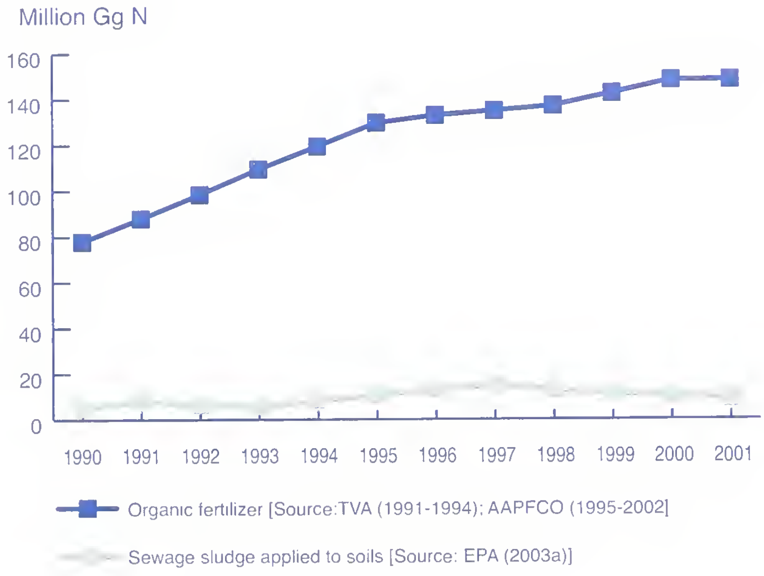
Source: TVA (1991-1994); AAPFCO (1996-2002)

Figure 3-7(B)
Livestock manure applied, 1990-2001



Source: EPA (2003a)

Figure 3-7(C)
Commercial organic fertilizer and sewage sludge applied, 1990-2001



direct and indirect emissions totaling 1.4 Tg CO_2 eq. in 2001. While small in overall magnitude, 2001 levels of N_2O from sewage sludge were 90 percent higher than 1990 levels (Table 3-4). Over this time period, more sludge was applied to land rather than disposing of it in landfills, dumping it into oceans, or burning (EPA 2003a).

3.10 Methods for Estimating N_2O Emissions from Agricultural Soils

Emissions of N_2O from nitrogen additions to cropland soils and cultivation of histosol soils are source categories analogous to those covered in Agricultural Soil Management in the U.S. GHG Inventory (EPA 2003a), with one exception. The U.S. GHG Inventory includes in Agricultural Soils Management direct and indirect emissions of N_2O from livestock manure deposited on pasture, range, and paddock, while the USDA GHG Inventory includes this source under Livestock GHG Emissions. The methodology outlined below is excerpted from Annex N of the U.S. GHG Inventory with permission from EPA to detail the methodology for emission estimates for Agricultural Soil Management; it does not include the portion on N_2O emissions from livestock manure deposited on pasture, range, and paddock. Methods for this source are covered in Chapter 2 of this report.

N_2O emissions were derived using activity data and emissions coefficients applied to the activity data. Activity data are developed for each source of nitrogen, estimated in terms of total amounts of nitrogen added. For histosol cultivation, activity data are annual areas of histosol soils that are cultivated[12] (Table 3-5). The activity data are derived from statistics, such as fertilizer consumption data or livestock population data, and various factors used to convert these statistics to annual amounts of nitrogen, such as fertilizer nitrogen contents or livestock excretion rates.

Nitrogen additions to soils result in direct and indirect emissions. Indirect emissions account

[12] Histosols are soils with a high organic carbon content. All soils with more than 20 to 30 percent organic matter by weight (depending on the clay content) are classified as histosols (Brady and Weil 1999).

for nitrogen that volatilizes to the atmosphere as NH_3 and NO_x, and subsequently returns to soils through atmospheric deposition, enhancing N_2O production. Additional nitrogen is lost from soils through leaching and runoff, and enters groundwater and surface water systems, from which a portion is emitted as N_2O. These two indirect emission pathways are treated separately, although the activity data used, except for livestock manure, are identical. The activity data for livestock manure are different from those used in other calculations. Here, total livestock manure (i.e., the sum of applied manure and manure used as a livestock feed supplement) is used in the volatilization and deposition calculation; and livestock manure applied on soils (i.e., applied manure) in the leaching and runoff calculation. Indirect emissions from manure deposited in pasture, range, and paddock are addressed in Chapter 2.

Table 3-5 Cultivated histosol areas

	Temperate	Sub-tropical
	1,000 ha	
1990	432	192
1991	431	193
1992	429	194
1993	431	194
1994	433	195
1995	435	195
1996	437	196
1997	439	196
1998	441	197
1999	443	197
2000	445	197
2001	447	198

Source: USDA NRCS (2000); EPA (2003a)

3.10.1 Activity Data

The activity data for this component include: a) the amount of nitrogen in synthetic and organic commercial fertilizers that are applied annually, b) the amount of nitrogen in livestock manure that is applied annually through both daily spread operations and the eventual application of manure that had been stored in manure management systems[13], c) the amount of nitrogen in sewage sludge that is applied annually, d) the amount of nitrogen in the aboveground biomass of nitrogen-fixing crops that are produced annually, e) the amount of nitrogen in crop residues that are retained on soils annually, and f) the area of histosols cultivated annually.

3.10.2 Application of Synthetic and Organic Commercial Fertilizer

Annual commercial fertilizer consumption data for the United States were taken from annual publications of synthetic and organic fertilizer statistics (TVA 1991, 1992a, 1993, 1994; AAPFCO 1995, 1996, 1997, 1998, 1999, 2000b, 2002) and a recent Association of American Plant Food Control Officials (AAPFCO) database (AAPFCO 2000a). These data were manipulated in several ways to derive the activity data needed for the inventory. First, the manure and sewage sludge portions of the organic fertilizers were subtracted from the total organic fertilizer consumption data because these nitrogen additions are accounted for under

[13] This source is distinguished from livestock manure deposited on pasture, range, and paddock because of the intent of the manure application and the land use resulting in the emission. In this case, manure has been collected and stored in a livestock operation. Emissions caused by the manure while in a storage system are accounted for in Chapter 2 and are attributed to livestock operations. After manure is removed from the storage system and used as an amendment for crop production, the resulting emissions are covered under this section on cropland soils.

"manure application" and "sewage sludge application."[14] Second, the organic fertilizer data, which are recorded in mass units of fertilizer, had to be converted to mass units of nitrogen by multiplying by the average organic fertilizer nitrogen contents provided in the annual fertilizer publications. These nitrogen contents are weighted average values, so they vary from year to year (ranging from 2.3 percent to 3.9 percent over the period 1990 through 2001). The synthetic fertilizer data are recorded in units of nitrogen, so these data did not need to be converted. Lastly, both the synthetic and organic fertilizer consumption data are recorded in "fertilizer year" totals (i.e., July to June); therefore, the data were converted to calendar year totals. This was done by assuming that approximately 35 percent of fertilizer usage occurred from July to December, and 65 percent from January to June (TVA 1992b). July to December values were not available for calendar year 2001, so a "least squares line" statistical extrapolation using the previous 11 years of data was used to arrive at an approximate value. Annual consumption of commercial fertilizers synthetic and non-manure/non-sewage organic in units of nitrogen and on a calendar year basis are presented in Figure 3-7 (A) and (C).

3.10.3 Application of Livestock Manure

To estimate the amount of livestock manure nitrogen applied to soils, it was assumed that all of the manure produced by livestock would be applied to soils with two exceptions: (1) the portion of poultry manure that is used as a feed supplement for ruminants, and (2) the manure that is deposited on soils by livestock on pasture, range, and paddock. In other words, it is assumed that all of the managed manure, except the portion of poultry manure that is used as a feed supplement, is applied to soils. The amount of managed manure for each livestock type was calculated by determining the population of animals that were on feedlots or otherwise housed in order to collect and manage the manure. In some instances, the number of animals in managed systems was determined by subtracting the number of animals in pasture, range, and paddock from the total animal population for a particular animal type. Annual animal population data for all livestock types, except horses and goats, were obtained for all years from the USDA NASS (Cattle: 2002, 2001, 2000, 1999, 1995; Cattle on Feed: 2002, 2001, 2000 ; Hogs and Pigs: 1998, 1994; Chicken and Eggs: 1998; Poultry Production and Value: 1999, 1995; Sheep and Goats: 1999 1994). Horse population data were obtained from the FAOSTAT database (FAO 2002). Goat population data for 1992 and 1997 were obtained from the Census of Agriculture (USDA NASS Census 1999); these data were interpolated and extrapolated to derive estimates for the other years. Information regarding poultry turnover (i.e., slaughter) rate was obtained from State NRCS personnel (Lange 2000). Additional population data for different farm size categories for dairy and swine were obtained from the Census of Agriculture (USDA NASS Census1999). Information regarding the percentage of manure handled using various manure management systems for dairy cattle, beef cattle, and sheep was obtained from communications with personnel from State NRCS offices, State universities, NASS, and other

[14] Organic fertilizers included in these publications are manure, compost, dried blood, sewage sludge, tankage, and "other." (Tankage is dried animal residue, usually freed from fat and gelatin). The manure and sewage sludge used as commercial fertilizer are accounted for elsewhere, so these were subtracted from the organic fertilizer statistics to avoid double counting.

experts (Poe et al. 1999, Anderson 2000, Deal 2000, Johnson 2000, Miller 2000, Milton 2000, Stettler 2000, Sweeten 2000, Wright 2000). Information regarding the percentage of manure handled using various manure management systems for swine, poultry, goats, and horses was obtained from Safley et al. (1992). A more detailed discussion of manure management system usage is provided in Chapter 2 of this report and Annex M of the U.S. GHG Inventory.

Once the animal populations for each livestock type and management system were estimated, populations were multiplied by an average animal mass constant (USDA NRCS 1996, USDA NRCS 1998, ASAE 1999, Safley 2000) to derive total animal mass for each animal type in each management system. Total Kjeldahl nitrogen[15] excreted per year for each livestock type and management system was then calculated using daily rates of nitrogen excretion per unit of animal mass (USDA NRCS 1996, ASAE 1999). The total poultry manure nitrogen in managed systems was reduced by the amount assumed used as a feed supplement (i.e., 4.2 percent of the managed poultry manure; Carpenter 1992). The annual amounts of Kjeldahl nitrogen were then summed over all livestock types and management systems to derive estimates of the annual manure nitrogen applied to soils and are shown in Figure 3-7 (B).

3.10.4 Application of Sewage Sludge

Estimates of annual nitrogen additions from land application of sewage sludge were derived from periodic estimates of sludge generation and disposal rates that were developed by EPA. Sewage sludge is generated from the treatment of raw sewage in public or private wastewater treatment works. Based on a 1988 questionnaire returned from 600 publicly owned treatment works (POTWs), the EPA estimated that 5.4 million metric tons of dry sewage sludge were generated by POTWs in the United States in that year (EPA 1993a). Of this total, 33.3 percent was applied to land, including agricultural applications, compost manufacture, forestland application, and the reclamation of mining areas. A subsequent EPA report (EPA, 1999) compiled data from several national studies and surveys, and estimated that approximately 6.7 and 6.9 million metric tons of dry sewage sludge were generated in 1996 and 1998, respectively, from all treatment works, and projected that approximately 7.1 million metric tons would be generated in 2000. The same study concluded that 60 percent of the sewage sludge generated in 1998 was applied to land (based on the results of a 1995 survey), and projected that 63 percent would be land applied in 2000. These EPA estimates of sludge generation and percent applied to land were linearly interpolated to derive estimates for each year in the 1990-2000 period. To estimate annual amounts of nitrogen applied, the annual amounts of dry sewage sludge applied were multiplied by an average nitrogen content of 3.3 percent (Metcalf and Eddy, Inc. 1991). Estimates for the year 2001 were held constant at the year 2000 level, as no new data were available (Bastian, 2002). Final estimates of annual amounts of sewage sludge nitrogen applied to land are presented in Figure 3-7 (C).

[15] Total Kjeldahl nitrogen is a measure of organically bound nitrogen and ammonia nitrogen in both the solid and liquid wastes.

3.10.5 Production of Nitrogen-Fixing Crops

Annual production statistics for beans, pulses, and alfalfa were taken from crop production reports (USDA NASS: Field Crops 1994 and 1998; Crop Production 2000, 2001, and 2002). Annual production statistics for the remaining nitrogen-fixing crops (i.e., the major non-alfalfa forage crops, specifically red clover, white clover, birdsfoot trefoil, arrowleaf clover, and crimson clover) were derived from information in a book on forage crops (Taylor and Smith 1995, Pederson 1995, Beuselinck and Grant 1995, Hoveland and Evers 1995), and personal communications with forage experts (Cropper 2000, Evers 2000, Gerrish 2000, Hoveland 2000, and Pederson 2000). The production statistics for beans, pulses, and alfalfa were in tons of product, which needed to be converted to tons of aboveground biomass nitrogen. This was done by multiplying the production statistics by 1 plus the aboveground residue-to-crop product mass ratios, dry matter fractions, and nitrogen contents. The residue to crop product mass ratios for soybeans and peanuts and the dry matter content for soybeans, were obtained from Strehler and Stützle (1987). The dry matter content for peanuts was obtained through personal communications with Ketzis (1999). The residue-to-crop product ratios and dry matter contents for the other beans and pulses were estimated by taking averages of the values for soybeans and peanuts. The dry matter content for alfalfa was obtained through personal communications with Karkosh (2000). The IPCC default nitrogen content of 3 percent (IPCC/UNEP/OECD/IEA 1997) was used for all beans, pulses, and alfalfa.[16] The production statistics for the non-alfalfa forage crops were derived by multiplying estimates of areas planted by estimates of annual yields, in dry matter mass units. These derived production statistics were then converted to units of nitrogen by applying the IPCC default nitrogen content of 3 percent (IPCC/UNEP/OECD/IEA 1997). The final estimates of annual aboveground biomass production, in units of nitrogen, are in Appendix Table B-9. The residue to crop product mass ratios and dry matter fractions used in these calculations are presented in Appendix Table B-8.

3.10.6 Retention of Crop Residue

It was assumed that 90 percent of residues from corn, wheat, barley, sorghum, oats, rye, millet, soybeans, peanuts, and other beans and pulses are left on the field after harvest (e.g., rolled into the soil, chopped and disked into the soil, or otherwise left behind) (Karkosh 2000).[17] It was also assumed that 100 percent of unburned rice residue is left on the field.[18] The derivation of residue nitrogen activity data was very similar to the derivation of nitrogen-fixing crop activity data. Crop production statistics were multiplied by aboveground residue to crop product mass

[16] This nitrogen content may be an overestimate for the residue portion of the aboveground biomass of the beans and pulses. Also, the dry matter fractions used for beans and pulses were taken from literature on crop residues, and so may be underestimates for the product portion of the aboveground biomass.

[1] Although the mode of residue application would likely affect the magnitude of N_2O emissions, an emission estimation methodology that accounts for this has not been developed.

[18] Some of the rice residue may be used for other purposes, such as for biofuel or livestock bedding material. Research to obtain more detailed information regarding final disposition of rice residue, as well as the residue of other crops, will be undertaken for future inventories.

ratios, residue dry matter fractions, residue nitrogen contents, and the fraction of residues left on soils. Annual production statistics for all crops except rice in Florida were taken from USDA NASS (Field Crops1994 and 1998; Crop Production: 2001 and 2002). Production statistics for rice in Florida were estimated by applying an average rice crop yield for Florida (Smith 2001) to annual Florida rice acreages (Schueneman 1999, 2001, Deren 2002).

Residue to crop product ratios for all crops were obtained from, or derived from, Strehler and Stützle (1987). Dry matter contents for wheat, rice, corn, and barley residue were obtained from Turn et al. (1997). Soybean and millet residue dry matter contents were obtained from Strehler and Stützle (1987). Peanut, sorghum, oat, and rye residue dry matter contents were obtained through personal communications with Ketzis (1999). Dry matter contents for all other beans and pulses were estimated by averaging the values for soybeans and peanuts. The residue nitrogen contents for wheat, rice, corn, and barley are from Turn et al. (1997). The nitrogen content of soybean residue is from Barnard and Kristoferson (1985), the nitrogen contents of peanut, sorghum, oat, and rye residue are from Ketzis (1999), and the nitrogen content of millet residue is from Strehler and Stützle (1987). Nitrogen contents of all other beans and pulses were estimated by averaging the values for soybeans and peanuts. Estimates of the amounts of rice residue burned annually were derived using information obtained from agricultural extension agents in each of the rice-growing States (see methods for Agricultural Residue Burning for more detail). The final estimates of residue retained on soil, in units of nitrogen (N), are in Appendix Table B-10. The residue to crop product mass ratios, residue dry matter fractions, and residue nitrogen contents used in these calculations are in Appendix Table B-8.

3.10.7 Cultivation of Histosols

Estimates of the areas of histosols cultivated in 1982, 1992, and 1997 were obtained from the USDA's 1997 National Resources Inventory (USDA NRCS 2000, as extracted by Eve 2001, and revised by Ogle 2002). These areas were grouped by broad climatic region using temperature and precipitation estimates from Daly et al. (1994, 1998), and then further aggregated to derive a temperate total and a sub-tropical total. These final areas were then linearly interpolated to obtain estimates for 1990 through 1996, and linearly extrapolated to obtain area estimates for 1998 through 2001 (Table 3-5).

3.10.8 Direct N_2O Emissions From Cropland Soils

Direct N_2O emissions from nitrogen additions and histosol cultivation were calculated separately, and each approach is discussed below. To estimate these direct emissions from nitrogen additions, the amounts of nitrogen applied were each reduced by IPCC values for fraction of nitrogen that is assumed to volatilize, the unvolatilized amounts were then summed, and the total unvolatilized nitrogen was multiplied by an emission factor of 0.0125 kg N2O-N/ kg Nitrogen (IPCC/UNEP/OECD/IEA 1997). The volatilization assumptions inherent in this approach are described below:

- Application of synthetic and organic commercial fertilizer: The total amounts of nitrogen applied in the form of synthetic commercial fertilizers and non-manure/non-sewage organic commercial fertilizers were reduced by 10 percent and 20 percent, respectively, to account for the portion that volatilizes to NH_3 and NO_x (IPCC/UNEP/OECD/IEA 1997).
- Application of livestock manure: The total amount of livestock manure nitrogen applied to soils was reduced by 20 percent to account for the portion that volatilizes to NH_3 and NO_x (IPCC/UNEP/OECD/IEA 1997).
- Application of sewage sludge: The total amount of sewage sludge nitrogen applied to soils was reduced by 20 percent to account for the portion that volatilizes to NH_3 and NO_x (IPCC/UNEP/OECD/IEA 1997, IPCC 2000).
- Production of nitrogen-fixing crops: None of the nitrogen in the aboveground biomass of nitrogen-fixing crops was assumed to volatilize.
- Retention of crop residue: None of the nitrogen in retained crop residue was assumed to volatilize.

To estimate annual N_2O emissions from histosol cultivation, the temperate histosol area was multiplied by the IPCC default emission factor for temperate soils (8 kg N_2O-N/ha cultivated; IPCC 2000), and the sub-tropical histosol area was multiplied by the average of the temperate and tropical IPCC default emission factors (12 kg N_2O-N/ha cultivated; IPCC 2000).

3.10.9 Indirect N_2O Emissions Induced by Applications of Nitrogen

To estimate N_2O emission from volatilization, the amounts of commercial fertilizer nitrogen and sewage sludge nitrogen applied, and the total amount of manure nitrogen produced, were each multiplied by the IPCC default fraction of nitrogen that is assumed to volatilize to NH_3 and NO_x (10 percent for synthetic fertilizer nitrogen; and 20 percent for nitrogen in organic fertilizer, sewage sludge, and livestock manure). Next, the volatilized amounts of nitrogen were summed, and then the total volatilized nitrogen was multiplied by the IPCC default emission factor of 0.01 kg N20-N/kg N (IPCC/UNEP/OECD/IEA 1997).

To estimate indirect emissions from leaching and runoff, the amounts of commercial fertilizer nitrogen and sewage sludge nitrogen applied, and the total amount of manure nitrogen applied, were each multiplied by the IPCC default fraction of nitrogen that is assumed to leach and runoff (30 percent for all nitrogen). Next, the leached/runoff amounts of nitrogen were summed, and then the total nitrogen was multiplied by the IPCC default emission factor of 0.025 kg N_2O-N/kg K (IPCC/UNEP/OECD/IEA 1997).

3.11 Uncertainty in Estimating N_2O Emissions from Agricultural Soils

The following discussion of uncertainty in estimating GHG emissions from cropland soils is modified from information in the U.S. GHG Inventory and reproduced here with permission from EPA.

The magnitude of N_2O emissions from cropland soils depends on nitrogen inputs and soil characteristics such as organic carbon availability, O_2 partial pressure, soil moisture content, pH, soil temperature, and soil amendments. The interacting impact of these variables on N_2O flux is complex and highly uncertain. Therefore, the IPCC default methodology, which is used here, is based only on N inputs and does not consider soil characteristics. This generalized approach treats all soils, except cultivated histosols, the same.

IPCC default emission factors for N_2O have associated ranges, the magnitudes of which indicate the uncertainty in the emission estimates. Most emission factor ranges are an order of magnitude, or larger. Developing a method to explicitly consider all driving factors of N_2O emissions will require more research; a prototype approach is described in Box 3-1, making use of process models to estimate N_2O emissions.

Uncertainties derive from activity data used to derive emission estimates. Activity data were often extrapolated from periodic surveys and commercial sales data, or, in some cases, they were based on expert opinion. For example: fertilizer statistics used in these estimates only include organic fertilizers in the commercial market; livestock excretion values were derived using simplifying assumptions concerning the types of management systems employed; and annual production and application estimates for sewage sludge were based on figures and projections calculated from surveys, yielding uncertainty levels as high as 14 percent (Bastian 1999). In addition, expert judgment was used to estimate the amount of residues left on soils, and the area of cultivated histosol soils was extrapolated from periodic data collected by a natural resource inventory, which was not explicitly designed as a soil survey. Production statistics for nitrogen-fixing forage legumes are not available except in the case of alfalfa; even so the statistics include alfalfa mixtures. Finally, conversion factors for nitrogen-fixing crops are based on a limited number of studies.

3.12 Mitigating N_2O and CH_4 Emissions from Cropland Agriculture

3.12.1 N_2O From Agricultural Soils

Fertilizer nitrogen (N) use efficiency in agricultural systems is limited by large losses of N through leaching and transformation to gaseous forms of N-oxides (ammonia, N_2O, nitric oxide, dinitrogen). In general, N-oxide emissions from mineral and organic soils can be reduced by management practices optimizing crop use of available N and minimizing losses from the soil-plant system. Strategies to increase overall N efficiency can consequently decrease N-oxide production, including N_2O emissions (Mosier et al. 1998a).

Sixty-nine percent of N_2O emissions from cropland soils are direct emissions, while the remaining 31 percent are emitted indirectly through volatilization and runoff. Management options focus on the direct sources of emissions; however, decreases in external N additions to crops will decrease both direct and indirect N_2O emissions. Current research, development, and

deployment efforts in the area of nutrient management focus on the following areas:

- *Precision agriculture* – targeted application of fertilizers, water, and pesticides.
- *Cropping system models* – tools to assist farm management decisions.
- *Controlled release of fertilizers and pesticides* – delivery of nutrients and chemicals to match crop demand and timing of pest infestation.
- *Soil microbial processes* – use of biological and chemical methods to manipulate microbial processes to increase efficiency of nutrient uptake, suppress N_2O emissions, and reduce leaching.
- *Agricultural best management practices* – limit N-gas emissions, soil erosion, and leaching.
- *Soil conservation practices* – utilizing conservation buffers and reserves.
- *Livestock manure utilization* – development of mechanisms to more effectively use livestock manure in crop production.
- *Plant breeding* – breeding varieties to increase nutrient use efficiency and decrease demand for pesticides, thus conserving energy.

3.12.2 CH_4 Emissions From Rice Fields

The amount of CH_4 emissions depends primarily on the area of rice cultivation. Since 1996, rice acreage has been controlled mainly by spring commodity prices and weather. Continuously flooded rice production is the most common production technique in the United States. Given the comparatively low level of emissions from rice paddies in the United States, there are no programs directed at limiting CH_4 emissions from rice fields. However, available research in the United States and internationally suggests that the most viable CH_4 reduction methods are cultivar selection, water management, and nutrient management.

3.13 Carbon Stock Changes in Agricultural Soils

Except for cultivated organic soils and liming practices, agricultural soils in the United States were estimated to accumulate about 59 Tg CO_2 eq. in 2001 (Table 3-6).[19] Much of the carbon change is attributable to the Conservation Reserve Program, land use conversions between annual croplands and

Table 3-6 Net carbon dioxide flux from agricultural soils, 1990, 1995-2001

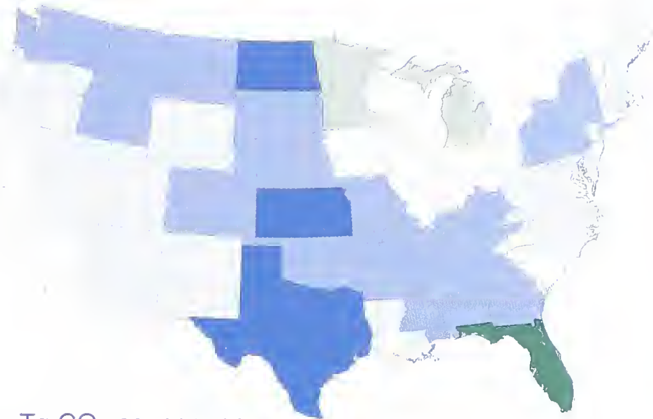
	1990	1995	1996	1997	1998	1999	2000	2001
				Tg CO_2 eq.				
Mineral Soils	(57.1)	(58.6)	(57.3)	(57.4)	(55.8)	(55.7)	(57.3)	(59.1)
Organic Soils	34.3	34.8	34.8	34.8	34.8	34.8	34.8	34.8
Liming of Soils	9.5	8.9	8.9	8.7	9.6	9.1	8.8	9.1
Total	(13.3)	(14.9)	(13.6)	(13.9)	(11.5)	(11.9)	(13.8)	(15.2)

Note: Parentheses indicate net sequestration.
Shaded values based on a combination of historical data and projections. All other values are based on historical data only.

[19] Emissions and sinks of carbon in agricultural soils are expressed in terms of CO_2 equivalents; carbon sequestration is a result of changes in stocks of carbon in soils, from which CO_2 fluxes are inferred. Units of CO_2 equivalent can be converted to carbon using a multiplier of 0.272.

perennial hay and grazing lands, and land management. Practices such as the adoption of conservation tillage, including no-till, which have taken place over the past two decades, and reduced frequency of summer-fallow are important drivers of carbon stock changes. Manure applications to cropland and pasture also impact the estimated carbon stock increase.

Map 3-1
CO_2 emissions and sequestration in agricultural soils, 1997



Note: Negative values correspond to sequestration.

In contrast, the small area of cultivated organic soils—less than 1 million hectares of a total 386 million hectares of agricultural land—concentrated in Florida, California, the Gulf and Southeastern coastal region and parts of the upper Midwest, was a net source of CO_2 emissions for all years covered by the inventory (1990-2001). About 35 Tg CO_2 eq. was emitted from cultivation of these soils in 2001 (Table 3-6). In addition, liming of agricultural soils resulted in emissions of about 9 Tg CO_2 eq. Total net carbon sequestration in 2001 was about 15 Tg CO_2 eq. when all of the above components are taken into consideration.

Carbon uptake on agricultural soils increased 14 percent (1.9 Tg CO_2 eq.) between 1990 and 2001 (Table 3-6). This trend is a function of increases in carbon uptake on mineral soils and decreases in carbon losses from liming, with slight increases in carbon losses on organic soils. Carbon uptake in mineral soils increased by 2 Tg CO_2 eq (3.5 percent), emissions from liming decreased slightly (<0.4 percent), and carbon losses on organic soils increased by less than 1 Tg CO_2 eq. (1.5 percent).

The remainder of this section focuses on carbon stock changes in mineral and organic soils. National estimates of carbon stock changes in mineral and organic agricultural soils are derived from several data sources including the Natural Resources Inventory (NRI), which is conducted every 5 years (USDA NRCS 2000). NRI data from 1982 through 1997 were used to derive carbon sock changes resulting from transitions in land use and management occurring over this time period.

State-level estimates were developed to provide a finer scale of information on trends in agricultural carbon stock changes. The national estimates for 1997 were disaggregated at the

Table 3-7 Areas in each land-use and management system for all U.S. land areas categorized as an agricultural use in 1992 or 1997 in the NRI

IPCC Land Use/Management Categories	1982	1992	1997
	Million hectares		
Medium-input cropping	87.49	77.17	78.27
High-input cropping (hay or legumes in rotation, winter cover crop, irrigated)	22.21	22.02	21.74
Low-input cropping (fallow, low residue crops)	30.96	28.92	25.13
Rice	2.71	2.13	2.22
Uncultivated land (hayland, rangeland, pasture, forest, federal)	210.04	207.77	210.26
Improved land (pasture or hayland with legumes or irrigation, continuous perennial crops)	31.19	33.65	31.43
CRP (set-aside)	0	13.78	13.23
Urban, water, miscellaneous non-cropland	1.78	0.96	4.11
Total	386.39	386.39	386.39

Source: USDA NRCS (2000).

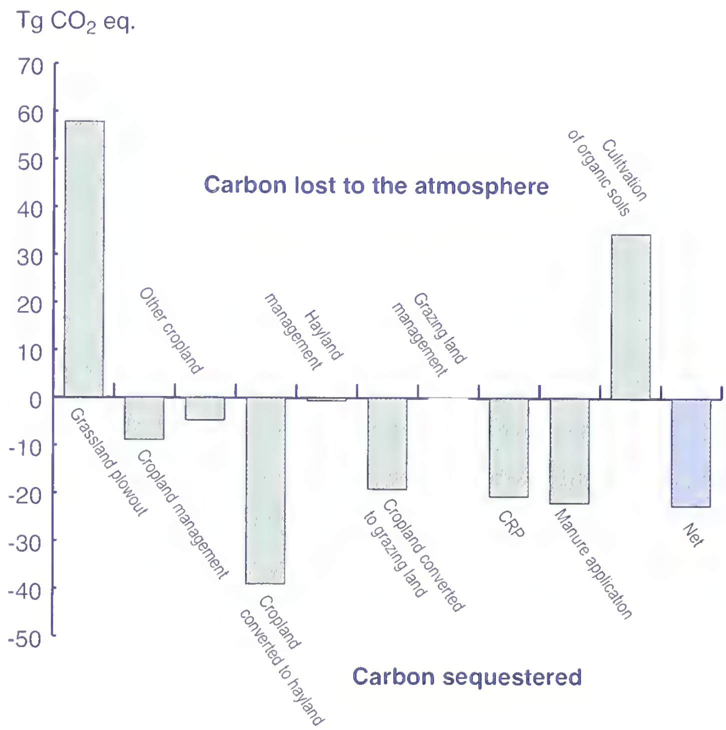
level of Major Land Resource Area (MLRA) and used to derive an approximate State-level estimate of carbon stock changes. State totals were computed using the values for all MLRAs occurring in the State, weighted by the share of the total State area comprised by each MLRA. Underlying trends in carbon stock changes are discussed below, focusing on individual components of the net soil carbon estimates (e.g., land set aside under CRP, manure applications, tilling practices, land use conversions between annual and perennial systems) and providing information on the magnitude of each component at the State level.

Given the complexity of illustrating the full range of carbon stock changes associated with each transition over the entire time sequence, the State-level analysis presented below shows net changes in agricultural soil carbon according to the major land use or management categories to which they belonged in 1997, which is the most recent year for which NRI data are available. For example, all lands that were in CRP as of 1997 are grouped together, irrespective of their prior management in earlier NRI data sets. Therefore, changes in aggregate carbon stocks for land classified as CRP in 1997 are a function of all changes in cropping system/land use that occurred after 1982, including the most recent change into CRP.

While total areas of annual cropland, hayland, and pasture are relatively constant over the inventory period, the NRI data indicate a substantial area of hayland and pasture was converted to annual cropping between 1982 and 1997 and a roughly corresponding area of annual cropland was put into hay land and pasture (Table 3-7). To better illustrate the effects of this land use "switching," data are provided separately for cropland and hayland/pasture, showing the gross effects of the land use change on soil carbon stocks, along with changes in carbon stocks occurring on land areas that underwent this "switching" of land use during the inventory period.

Net carbon stock changes in agricultural soils are shown for each State in Map 3-1 for 1997. These net changes are comprised of the sum of changes in individual components both in positive (carbon uptake) and negative (carbon losses to the atmosphere) directions (Appendix Table B-11). In 1997, crop cultivation resulted in carbon losses to the atmosphere on the order of 93 Tg CO_2 eq., driven mainly by losses following conversion of pasture and hayland into annual cropping systems (emissions of 58 Tg CO_2 eq.) and by losses from the cultivation of carbon-rich organic soils (emissions of 35 Tg CO_2 eq.) (Figure 3-8). However, these losses were more than offset by land use conversions from cultivated cropland to hay and grazing lands, which sequestered close to 60 Tg CO_2 eq. of carbon in soils. Lands enrolled in the Conservation Reserve Program and the application of manure fertilizer to crop and grazing lands provided further carbon uptake in soils of about 40 Tg CO_2 eq. The net effect of these land management actions and land use conversions is the sequestration of about 23 Tg CO_2 eq. of carbon in agricultural soils.

Figure 3-8
Components of CO_2 emissions and sequestration in agricultural soils, 2001



Land set aside from production into the Conservation Reserve Program is an important contributor to soil carbon stock increases (Figure 3-8), with the highest amounts registered for States where there are large areas of CRP enrollment (e.g., North Dakota, South Dakota, Texas, Kansas, Montana, Colorado) (Appendix Table B-11). Most States show small gains on long-term annual cropland, due to increased use of more carbon-conserving practices such as conservation tillage (0.15 Tg CO_2 eq. in Colorado; 0.48 Tg CO_2 eq. in Illinois; 0.66 Tg CO_2 eq. in Nebraska; 0.59 Tg CO_2 eq. in Texas) (Appendix Table B-11). Manure additions at the State level contributed to carbon sequestration in soils of 0.5 Tg CO_2 eq. on average, with State levels ranging from no impact of manure application to sequestration levels of nearly 2 Tg CO_2 eq. The largest gains from manure application were in the major livestock-producing States (e.g., California, Texas, Wisconsin, Iowa, North Carolina, Nebraska) (Appendix Table B-11). CO_2 emissions from cultivated organic soils are restricted to a few States having significant areas of

those soils. Nearly one-third of the national total occurs in Florida. Other major States include California and several States in the Great Lakes region (e.g., Minnesota, Michigan, Wisconsin, and Indiana).

3.14 Methods for Estimating Carbon Stock Changes in Agricultural Soils

Researchers at the Natural Resources Ecology Laboratory (NREL) at Colorado State University developed the methods and provided estimates for this chapter (Ogle et al. 2003). The estimates were originally developed for the U.S. GHG Inventory following an approach modified from the Intergovernmental Panel on Climate Change (IPCC) Guidelines for National Greenhouse Gas Inventories (IPCC/UNEP/OECD/IEA 1997). The approach was expanded for this report to include State-level estimates of carbon stock changes. The analysis was designed first and foremost to inform policymakers at the national level, using information derived from the U.S. national-level greenhouse gas inventory. The objectives of this report are to provide a preliminary assessment of changes in soil carbon emissions/sinks at the State level.

3.14.1 Stock Change Calculations for Mineral Soils

For mineral soils, the method estimates effects of land use change and management on relative stock changes over a defined time interval (default of 20 years) and soil depth (default of 30 cm). The method assumes that, in the absence of any changes in management or land use over the 20-year inventory period, soil carbon stocks do not change. In essence, they are assumed to be in an equilibrium or steady-state condition. There are three main kinds of information necessary to apply the method: 1) stock change factors for specific land use and management practices, 2) reference carbon stocks for the baseline agricultural condition (i.e., conventionally tilled cropland with row-cropping rotations) to which the stock change factors are applied, and 3) activity data that record the changes in land use and management over time. These are combined in the following way:

$$\Delta C = [\ \textstyle\sum(SC_{it} - SC_{i(t-20)}) * LA_i\ ] / 20$$

$$SC_{it} = SC_R * F_{LU} * F_T * F_I$$

where SC_i is soil organic carbon stock for the ith parcel of land at time t and t-20 years, LA_i is land area of each parcel, SC_R is the reference carbon stock and F_{LU} , F_T , F_I are stock change factors (for land-use type, tillage regime, and carbon input level, respectively), which define the land use and management conditions on each parcel of land. Land area parcels represent the areas associated with each type of land use/management system (as defined by the stock change factors), stratified by climate and soil type.

3.14.2 Climate Zones and Reference Soil Carbon Stocks

Stock change factors and reference carbon stocks can vary for different climate regimes and soil types. The IPCC method defines eight climate types according to mean annual temperature,

precipitation, and potential evapotranspiration. Six of these occur in the continental United States. The PRISM (Parameter-elevation Regressions on Independent Slopes Model) long-term monthly climate data set (Daly et al. 1998) was used to classify each of the 180 Major Land Resource Areas (MLRAs) in the United States into climate zones. MLRAs were chosen as the main spatial entity for the national estimates because each MLRA represents a geographic unit with relatively similar soils, climate, water resources, and land uses (NRCS 1981) and the management/land use activity data used (see below) could be readily aggregated by MLRA.

Reference soil carbon stocks (SC_R) were stratified by climate region and categorized into six major groupings, based on taxonomic orders that relate to soil development and physical characteristics that influence soil carbon contents. Estimates for carbon stocks under conventionally managed cropland (defined as the reference land use) were derived from the National Soil Survey Characterization Database (USDA NRCS 1997)

3.14.3 Land Use and Management Factor Values

Management factors (i.e., F_{LU}, F_T, F_I) representative of U.S. conditions were estimated from published studies (Ogle et al. 2003). The factors quantify the relative carbon stock change resulting from changing land use and management, including tillage practices, cropping rotation or intensification, and land conversions (including set-asides in the Conservation Reserve Program). Studies from the United States or Canada were used that met a minimum set of criteria, including reporting of SOC stocks (or information to compute stocks), depth of sampling, time, and management treatments. Factors were estimated for the effect of management practices at 20 years for the top 30 cm of the soil profile (Appendix Table B-13).

3.14.4 Land Use and Management Activity Data

Land use and management data were based primarily on the National Resources Inventory (NRI) (USDA NRCS 2000). The NRI represents a robust statistical sampling of land use and management on all non-Federal land in the United States, and greater than 400,000 NRI survey points occurred in agricultural lands and were used in the inventory analysis. Among the information in the NRI are land use category, soil description and, for agricultural lands, crop type. Based on the NRI, crop management systems were aggregated into 22 different categories. Land areas grouped by major land use and management system types are shown in Appendix Table B-14.

Tillage practices are not included in the NRI. Thus, supplemental data were used from the Conservation Technology Information Center (CTIC 1998), which reports tillage practices by major crops and county on an annual basis (Appendix Table B-15). CTIC data do not differentiate between continuous and intermittent use of no-tillage, which is important for estimating SOC storage. Thus regional-based estimates for continuous no-tillage (defined as 5 or more years of continuous use) were derived based on consultation with a CTIC expert (D. Towery 2001, personal communication).

Other data used to supplement the NRI included: (1) area of cropland restored to wetland and (2) manure applications to crop and grazing land. Data were available for wetland restoration under the CRP program for the Northern Prairie Pothole Region (Euliss and Gleason 2002, personal communication), including the amount of area restored and estimates of carbon stock change over tin For manure (and sludge) application, EPA compiled data on the amount of total manure produced an available for application to agricultural land (EPA 2003a). Since data on actual application rates wer not available, manure was assumed to be applied according to recommended rates based on the assimilative capacity for crops (Kellogg et al. 2000). Supplemental data are available regarding the amount of cropland area receiving manure and sewage sludge for major crops in the United States (USDA ERS 2000). The percentage of fields receiving manure and sewage sludge had been estimate between 1990 and 1997 for corn, soybeans, winter wheat, cotton, and potatoes. This information wa used in conjunction with the data collected from the USDA NASS Agricultural Statistics database (http://www.nass.usda.gov:81/ipedb/), which provides information on the amount of land planted to each crop, for estimating the cropland area receiving manure and sewage sludge. The remaining area receiving manure and sewage sludge was assumed to occur in grazing lands (calculated as the difference between the total area receiving manure and sewage sludge and the cropland area receivin manure and sewage sludge).

3.14.5 Carbon Flux from Organic Soils

Organic soils (i.e., peat, mucks) that have been drained and converted to cropland or pasture use are subject to potentially high rates of carbon loss. Carbon loss rates (mean and variance) were estimate based on field studies in the United States and Canada stratified by the three major temperature regin characterized in the IPCC methodology (Appendix Table B-16). However, based on the limited fiel studies there was no significant difference between mean rates for the warm temperate and subtropic regions. Carbon loss from cultivated organic soils is based primarily on measurements of subsidence of the land surface over time, corrected for the relative contributions of compaction and erosion, to derive an estimate of the losses due to decomposition and CO_2 emission. The most recent data available from cultivated organic soils in Florida were included (D. Morris, personal communication resulting in a decrease in the estimated loss rates to about 14 tons C per hectare per year, from 20 ton C per hectare per year suggested by the IPCC. Flux rates were applied to area estimates for cultivate organic soils by climate region to estimate emissions.

3.15 Uncertainty in Estimating Carbon Stock Changes in Agricultural Soils

The IPCC-based methodology uses empirical models to estimate carbon stock changes in response to specific land use and management activities on agricultural lands. For estimating carbon gains it can be viewed as inherently conservative for two reasons. First, stock change factor values are statistical derived for a 20-year time period; hence, if carbon gains in response to a specific management chang are sustained over a longer time period, then gains beyond 20 years will be unaccounted for in the inventory.[20] Second, the IPCC method deals only with discrete changes in management systems and

[20] However, changes that result in decreases in C stocks, such as conversion of native vegetation or pasture to cultivated annual crops, are also derived for a 20-yr time frame, and thus losses sustained over a longer period of time would be underestimated.

thus longer term 'baseline' trends in productivity and carbon inputs to soil (i.e., independent of changes in tillage or crop rotation) are not accounted for in the present methodology. For example, several authors have suggested that the general increase in U.S. agricultural productivity (on the order of 1-2 percent annually for major crop species over the past 50 years) has contributed to soil carbon sequestration through a steady increase in crop residue production (e.g., Cole et al. 1993, Allmaras et al. 2000). Because this steady productivity increase is not readily characterized as a 'change' in land use or management, it is difficult to capture in the IPCC method as presently implemented. Finally, the IPCC-based method is currently applied at a highly aggregated level,[21] and thus may not capture many regional differences in how soil carbon stocks respond to management.

Carbon loss from cultivated organic soils is based primarily on measurements of subsidence of the land surface over time, corrected for the relative contributions of compaction and erosion, to derive an estimate of the losses due to decomposition and CO_2 emission. Few studies are available, leading to significant uncertainty in the estimates of carbon flux rates from organic soils. In addition, some of the published data are from older studies in which loss rates may be higher than they would be at present if better water management practices (i.e., maintaining higher water tables) were used.

Annex P of the U.S. GHG Inventory (EPA 2003a) contains details on the quantitative uncertainties associated with national estimates of CO_2 emissions and sinks in agricultural soils.

3.16 Alternative Approaches for Estimating Carbon Stock Changes

Alternative approaches using dynamic simulation models such as the Century model have been used to estimate changes in agricultural soil carbon emissions and sinks at national and regional scales (e.g., Paustian et al. 2002, Falloon et al. 2002, Brenner et al. 2001, 2002, Smith et al. 2002). Such dynamic models offer some advantages in that factors such as long-term trends in crop productivity, as well as finer scale representation of climate, soil, and management conditions, can more easily be incorporated. In addition, since they operate as a continuous simulation (as opposed to a discrete time period as in the IPCC method), they are better suited to producing annual estimates.

For example, dynamic simulation models were used to study agricultural soil carbon changes in Nebraska (Brenner et al. 2002). Data on climate, soils, land use, and management were collected for each county in the State, with assistance from local conservation district and USDA NRCS personnel. These data were used as inputs to the Century model to estimate soil carbon emissions and sinks from 1990 to 2000 and to make projections of potential changes in soil carbon with increased adoption of conservation practices. For comparison, the Century model study estimated that Nebraska agricultural soils in 1997 sequestered 4.62 Tg CO_2 eq. per

[21] Work is underway to develop a more disaggregated set of factor values to better reflect regional climate and soil influences on C stock responses to management.

year, compared to the State-level estimate of 1.98 Tg CO_2 eq. based on the IPCC method.

For some practices, such as the effect of the Conservation Reserve Program on carbon stock change, estimates were similar (0.92 Tg CO_2 eq. per year using Century and 0.99 Tg CO_2 eq. per year using IPCC). The Century analysis estimated that much of the total carbon gains in the State occurred on annual cropland under intensive and reduced tillage (0.92 and 2.42 Tg CO_2 eq. respectively), which comprises about 95 percent of the annual cropland area (ca. 20 million acres). Some of this gain is due to the long-term impact of steady increases in productivity as mentioned above, along with a shift towards less intensive tillage. An additional gain of 0.37 Tg CO_2 eq. per year was estimated for continuous no-till cropland, which comprised less than 1 million acres, bringing the total gain on annual cropland to 3.7 Tg CO_2 eq. per year using the Century approach.

In contrast, the IPCC method estimated a net sequestration increase on all cropland in Nebraska (excluding CRP) of only 1.1 Tg CO_2 eq. per year versus the 3.7 Tg CO_2 eq. per year estimated using Century. One reason for the discrepancy between the methods is that the Century analysis did not include the offsetting effects of "switching" between annual and perennial cropland (i.e., hayland, rangeland), which in the IPCC analysis accounted for a net loss of about 0.7 Tg CO_2 eq. per year. However, the overall increase in agricultural productivity in the United States over the past several decades likely represents some additional carbon sequestration that is not accounted for in the IPCC method. Two other State-level studies in Iowa and Indiana have been conducted using the same model-based approach. In both cases, the results were substantially higher estimates of carbon sequestration rates than in the IPCC-based approach, largely for the same reasons discussed above.

3.17 Summary and Recommendations

The present analysis builds on a simple method designed for national-level greenhouse gas inventories. It includes factors, estimated from U.S. field studies that encompass most of the major influences of land use and management on soil carbon stocks. The model is applied to data from the National Resources Inventory and other sources that provide a robust, statistically based estimate of agricultural management activities during the past two decades. However, the data and model components are most appropriate for use at broad scales, and thus their interpretation at sub-national levels should be viewed with caution. The State-level disaggregation presented here is intended to provide insight into possible regional trends, and allow comparison with other State-level components of agricultural greenhouse gas emissions.

Agriculture was estimated in 2001 to sequester about 59 Tg CO_2 eq. on mineral soils, which was partially offset by net losses of 33-37 Tg CO_2 eq. from cultivated organic soils, yielding a net of about 22 Tg CO_2 eq. (EPA 2003a). The IPCC method has also been applied to estimate *potential* soil carbon sequestration with adoption of best management practices in the United States, which yielded an estimate of about 367 Tg CO_2 eq. per year (Sperow et al. in 2003). This amount is similar to other independent estimates of potential soil carbon sequestration

rates of 275-623 Tg CO_2 eq. per year (e.g., Bruce et al. 1999, Lal et al. 1998). The differences between these potential and actual rates are indicative of the large 'room for improvement' that exists in implementing agricultural practices that enhance soil carbon storage. For example, while no-till practices are becoming more widely used, long-term continuous no-till is still practiced on only a few percent of the annual crop area within most regions. Intermittent use of no-till (such as no-till soybean and more intensive tillage of corn in a soybean-corn rotation) leads to much less carbon sequestration compared to continuous no-till. Similarly, much of the positive effects of changing crop rotations to include more hay, conversions to pasture, conservation set-asides, etc., that are occurring in some locations appear to be offset to varying degrees by other changes occurring in other locations, i.e., the "switching" effect described earlier. Thus, more consistent and sustained adoption of improved practices is needed to realize the full potential for carbon sequestration.

Methodologies suitable for more local assessment and management planning to support greenhouse gas mitigation are not yet available for all parts of the United States. However, work in some States, such as described earlier for Nebraska, shows promise in making available information and decision support aids that can be applied at field and local scales. Such an approach can capture changes in carbon stocks due to long-term trends in productivity that are not tied to specific management changes and also are more applicable for annual accounting purposes. Expanding dynamic model-based estimates to other States and to national GHG accounting purposes would enable a more comprehensive accounting of all the factors influencing soil carbon stocks changes. However, applying these more complex approaches to improve carbon accounting will require additional resources and research, including robust estimates of uncertainty.

Numerous data enhancements would be beneficial to improve inventory and carbon accounting estimates. Maintaining and expanding the utility of the National Resources Inventory is crucial because the NRI represents the only statistically valid, national-level source of information on land management and land resources. Collection of additional data at NRI points, such as tillage practices, residue management, organic amendments (type and rate; e.g., manure and sewage sludge), fertilization practices, and wetland restoration, would improve the input data and reduce uncertainty in the estimates. Other potential improvements include application of remote sensing technology to estimate the area distribution and changes over time in land use, crop rotation, and tillage practices. Such data could be overlain with soil maps using a GIS, to provide 'wall to wall' coverage of land management changes to complement the point sampling provided by NRI. Establishing a network of soil monitoring points, where carbon stocks could be re-measured over time along with records of management practices, either in conjunction with or complementary to the NRI, would provide a valuable means for verifying estimated carbon stock changes and further improving estimation methods.

4.1 Net Sequestration of CO_2 in Forest Ecosystems and Forest Products

Carbon dioxide is continually exchanged between forest ecosystems and the atmosphere as illustrated by the pools and fluxes in Figure 4-1. Photosynthesis leads to the conversion of carbon dioxide into organic carbon in growing plants, and some of the carbon thus sequestered as plant biomass is subsequently lost through respiration. A large net flux of carbon from atmosphere to tree accompanies early tree growth. Over time, the net rate of exchange decreases due to increasing carbon loss through respiration or the loss and subsequent decomposition of plant material as litter and woody debris. A large amount of carbon is released to the atmosphere as trees die and decompose. Other mechanisms of carbon loss from forest systems include physical removal of organic matter or rapid loss through natural disturbance, such as fire. A significant form of removal in the United States is harvest of wood, but carbon can also be removed through runoff or leaching through soil. Subsequent forest regeneration and growth can then reestablish the section of forest as a sink of atmospheric carbon dioxide. The continuous exchange of carbon with the atmosphere is by far the most significant role of forests in the U.S. greenhouse gas inventory; the net flux of carbon dioxide from the atmosphere to forest ecosystems and harvested wood products in 2001 is estimated at 759 Tg CO_2 eq. Thus, forest carbon budgets are the focus of this chapter.

Forest management is an activity involving the regeneration, tending, protection, harvest, and utilization of forest resources to meet goals defined by the forest landowner. Forest management regimes vary by forest ecosystem, landowner objectives, and economic possibilities. Increasing tree volume per area of forest generally increases carbon sequestration. Relatively passive management may include tree harvest and removal, followed by natural regeneration, or riparian area management that consists of consciously retaining a buffer strip of trees along a watercourse. Intensive management may consist of site preparation, improved stocking, species conversion, planting genetically improved stock, application of pesticides or fertilizer, and improvement cuttings such as thinning or precommercial thinning.

Goals of forest management can focus on one or more outcome, output, or benefit. Example benefits include conservation of soil, water, vegetation, wildlife, carbon, and nutrient resources. Forest management can enhance levels of carbon stocks. Although some practices may decrease carbon storage for a given site-age-type dynamic, generally more carbon may be sequestered in forest systems through improved forest management practices, afforestation, increased productivity, reduced conversion to non-forest uses, lengthened rotations in some systems, and increased proportion and retention of carbon in harvested wood products. Sustainable short-rotation woody crops systems offer the opportunity to rapidly deploy new, faster growing genetic material, sequester carbon in the soil, add to the wood products pool, and provide energy feedstocks as fossil fuel offsets. Afforestation offers significant opportunities to capture and store carbon on lands that are not currently forested. This is a particularly useful tool for marginal agricultural lands.

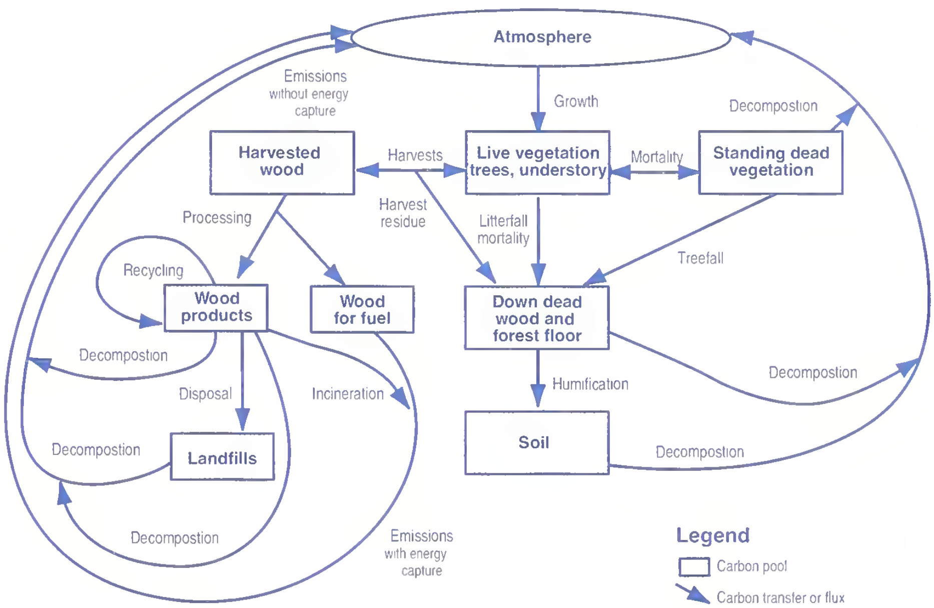
Improvements in the management of wood products in use and in landfills provide a number of

opportunities to reduce emissions and increase sequestration. Continuing development of wood products can increase their durability and recycling potential. This increases opportunities for sequestration in wood products and potentially expands the market for wood products relative to more emission-intensive materials.

Urban trees provide important offset opportunities. Advances in design and deployment of trees in urban environments can provide significant fossil fuel savings for heating and cooling, through microclimate management. Development of urban tree waste management and recycling processes and systems would reduce emissions and increase sequestration opportunities.

Figure 4-1

Summary diagram of forest carbon stocks and carbon transfer among stocks



Source: Adapted from USEPA (2003a), and Heath and others (2003)

This chapter summarizes carbon stocks and stock changes (net carbon fluxes) for U.S. forestland and wood products. Carbon stock estimates are based on extensive forest inventory data, and are consistent with internationally recognized methods for carbon accounting such as the *Revised 1996 IPCC Guidelines* (IPCC/UNEP/OECD/IEA 1997). Summaries of information included in this chapter represent an update of previous national forest carbon budgets (Birdsey 1992, Birdsey and Heath 1995) and were provided to the United States Environmental Protection Agency for the Land-Use Change and Forestry section of the *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2001*(EPA 2003a). Estimated carbon budgets are for the year 2001. Principal sources of forest inventory data are compilations of national forest resource statistics for 1987 and 1997 (Waddell and others 1989, Smith and others 2001). Estimates for years after 1997 are projections based on simulation modeling of expected land use change and forest growth, harvest, and utilization.

4.2 Method for Estimating Forest Carbon Mass

4.2.1 Carbon in Forest Ecosystems and Forest Products

Forest structure provides a convenient modeling framework for simulation models to estimate carbon stocks. Carbon stocks in forest ecosystems are modeled as five distinct pools, which are as follows:

- **Trees**: live and standing dead trees including coarse roots, stems, branches, and foliage (live trees only).
- **Understory vegetation:** including the roots, stems, branches, and foliage of seedlings (trees less than 2.5 cm diameter), shrubs, and bushes.
- **Forest floor:** including fine woody debris up to 7.5 cm diameter, tree litter, and humus.
- **Down dead wood:** including logging residue and other coarse dead wood on the ground and larger than 7.5 cm diameter, and stumps and coarse roots of stumps.
- **Soil:** including all organic material in soil excluding any carbon specified for above pools.

Such pools can also be grouped as live vegetation, standing-dead vegetation, the accumulation of dead material on the soil surface, and organic carbon in soil. This is convenient for modeling mechanisms that affect movement of carbon among pools, as illustrated in Figure 4-1. Transfer among pools depends on processes such as growth, mortality, decay, natural disturbances, and the anthropogenic activities of harvesting, thinning, clearing, and replanting. Carbon is continuously cycled through and among these storage pools and between forest ecosystems and the atmosphere.

The net change in forest carbon over an interval of time is not necessarily equal to the net flux between forests and the atmosphere because timber harvests may not result in an immediate release of harvested carbon to the atmosphere. Harvested wood carbon removed from the forest is summarized as two pools, wood products in use, and wood discarded in landfills. As wood products combust or decay over time, the carbon is ultimately reemitted to the atmosphere as carbon dioxide and methane.

The path by which carbon returns to the atmosphere can be important to overall carbon accounting of forest systems. Emissions can occur from wood burned for energy or from burning or decay of wood without energy capture (Figure 4-1). We include these two "pools" of emitted carbon dioxide—with and without energy capture—because they help provide a complete picture of forest carbon budgets. The rate of emission varies considerably among different product pools. For example, if timber is harvested for energy use, combustion results in an immediate release of carbon. Conversely, if timber is harvested and subsequently used as lumber in a house, it may be many decades or even centuries before the lumber decays and carbon is released to the atmosphere. If wood products are disposed of in landfills, the carbon contained in the wood may be released gradually over years or decades as carbon dioxide or

methane gas. Wood burned for energy, as a substitute for fossil fuel, is a relatively large pool, and there may be potential for even greater energy recovery from waste wood products.

4.2.2 Background Information in Forest Inventories

The Forest Inventory and Analysis Program (FIA) of the USDA Forest Service has conducted consistent forest surveys based on extensive statistically based sampling of much of the forestland in the United States since 1952. The United States has approximately 300 million hectares of forestland; about 250 million hectares are located in the conterminous 48 States and form the basis for the estimates provided in this chapter (Smith and others 2001). Seventy-nine percent of the 250 million hectares are classified as timberland, meaning they meet minimum levels of productivity and are available for timber harvest. Historically, timberlands in the conterminous 48 States have been more frequently and intensively surveyed than other forestlands. Of the remaining 51 million hectares, 16 million hectares are reserved forestlands (withdrawn by law from management for production of wood products) and 35 million hectares are lower productivity forestlands.

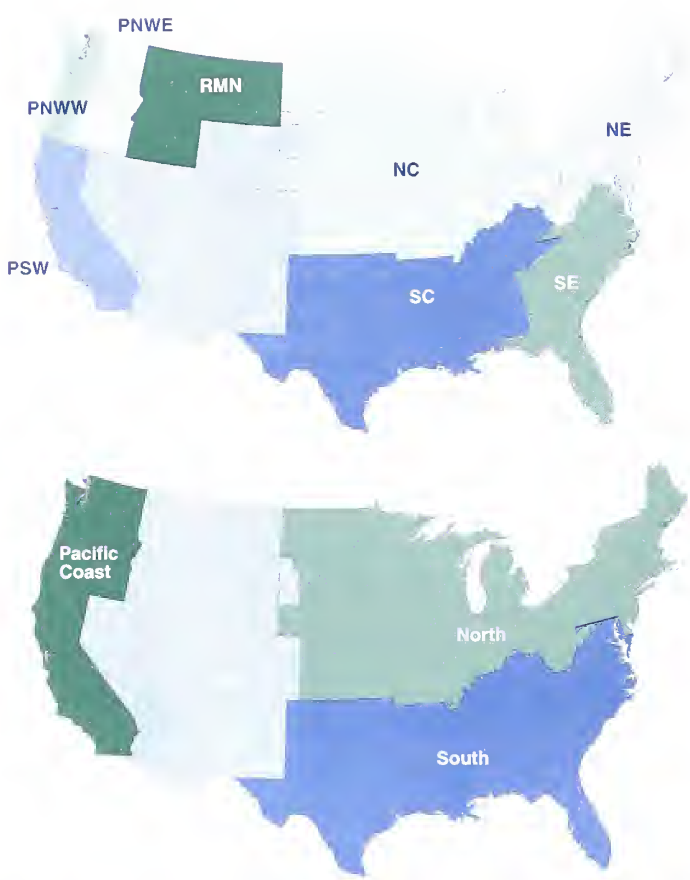
Historically, the main purpose of the FIA program has been to estimate areas, volume of growing stock, and timber products output and utilization factors. Growing stock is simply a classification of timber inventory that includes live trees of commercial species meeting specified standards of quality (Smith and others 2001). Timber products output refers to the production of industrial roundwood products such as logs and other round timber generated from harvesting trees, and the production of bark and other residue at processing mills. Utilization factors relate inventory volume to the volume cut or destroyed when producing roundwood (May 1998). Growth, harvest, land-use change, and other estimates of change are derived from repeated land, aerial, and satellite surveys. Although carbon sequestration is not directly measured in these surveys, the data provide a strong foundation for carbon estimates because the inventories collect data that are highly correlated with carbon stocks. Carbon stocks and fluxes can be estimated using mathematical models and supplemental data from research studies and more intensive monitoring sites.

Forest inventories used in this chapter are based on the State-by-State surveys periodically conducted (every 5-14 years, depending on the State) by regional FIA programs during the 1980s and 1990s. Compilations of these data for 1987 and 1997 are given in Waddell and others (1989) and Smith and others (2001), with trends discussed in the latter citation. FIA has adopted a new annualized design that includes a more extensive and nationally consistent database, which is also conducted State-by-State (Miles and others 2001, <http://www.ncrs.fs.fed.us/4801/fiadb/index.htm>). The annualized survey also includes plans for measurements to facilitate direct estimates of carbon in woody debris, forest floor, and mineral soil from systematically sampled data. However, these data are only beginning to become available from a limited number of States. One feature of the periodic survey data used in this chapter is that some forestlands were surveyed less often than others. Estimates of carbon stock changes can be influenced by the frequency of these surveys.

4.2.3 Estimating Forest Carbon Stocks and Stock Changes

Map 4-1
Regions used for carbon stock summaries

Regions in the upper map are: Pacific Northwest – Westside (PNWW); Pacific Northwest – Eastside (PNWE); Pacific Southwest (PSW); Rocky Mountain – North (RMN); Rocky Mountain – South (RMS); North Central (NC); Northeast (NE); South Central (SC); and Southeast (SE).



The inventory data are converted to carbon using conversion factors or using the model forest carbon budget simulation model FORCARB2 (Birdsey and Heath 1995, Heath and others 2003). The seven component carbon pools, or stocks, are listed in Table 4-1, and relationships among the parts are illustrated in Figure 4-1. Separate estimates are made for each of the forest ecosystem carbon pools: trees, understory vegetation, forest floor, down dead wood, and soils. Stock change, or net annual flux, is based on the difference between two successive estimates of stock divided by the number of years in the interval. Negative fluxes (in parentheses in tables) represent gains in carbon stocks, or removal of carbon dioxide from the atmosphere; this is the convention in all tables and figures. Note that the sign convention is maintained to indicate direction of carbon flux; in the text, direction of flux is indicated by explicitly identifying sources and sinks. Total forest ecosystem stocks or stock changes can be obtained from summing the five constituent pools: similarly, totals for carbon in harvested wood can be obtained from summing the two pools (Table 4-1). Projected carbon stock changes are derived from areas, volumes, growth, land-use changes, and other forest characteristics projected in a system of models (see Haynes and others 2003) representing the U.S. forest sector, including FORCARB2.

Table 4-1 Summaries of forest area, carbon stocks for 1987, 1997, and 2002; and average net annual stock change for the intervals 1987-1996 and 1997-2001

Year/Interval	Stock Jan. 1, 1987	Stock Change 1987-1996	Stock Jan. 1, 1997	Stock change 1997-2001	Stock Jan. 1, 2002
	Tg CO_2 eq.	*Tg CO_2 eq./yr*	*Tg CO_2 eq.*	*Tg CO_2 eq./yr*	*Tg CO_2 eq.*
Forests	174,398		182,092		184,822
Trees	55,579	(469)	60,272	(447)	62,508
Understory	1,642	(9)	1,733	(15)	1,806
Forest floor	15,536	(24)	15,778	29	15,631
Down dead wood	7,541	(54)	8,080	(59)	8,373
Forest soils	94,100	(213)	96,229	(55)	96,504
Harvested Wood	7,035		9,080		10,140
Wood products	4,342	(49)	4,833	(58)	5,123
Landfilled wood	2,693	(155)	4,247	(154)	5,016
Energy Capture		176		185	
Emitted		129		136	
	Tg C	*Tg C/yr*	*Tg C*	*Tg C/yr*	*Tg C*
Forests	47,595		49,695		50,440
Trees	15,168	(128)	16,449	(122)	17,059
Understory	448	(3)	473	(4)	493
Forest floor	4,240	(7)	4,306	8	4,266
Down dead wood	2,058	(15)	2,205	(16)	2,285
Forest soils	25,681	(58)	26,262	(15)	26,337
Harvested Wood	1,920		2,478		2,767
Wood products	1,185	(13)	1,319	(16)	1,398
Landfilled wood	735	(42)	1,159	(42)	1,369
Energy capture		48		51	
Emitted		35		37	

Area of forest	Jan. 1, 1987	Jan. 1, 1997	Jan. 1, 2002
1,000 ha	245,593	250,027	251,029

Parentheses indicate net sequestration.

Table 4-2 Forest carbon stocks, area, and net annual stock change by forest type, 2001

	Stocks			*Area*	*Net Annual Change*	
Forest type	**Biomass**	**Nonliving Plant Mass**	**Soil Organic Carbon**	**Area**	**Biomass**	**Nonliving Plant Mass**
	Tg CO_2 eq.	*Tg CO_2 eq.*	*Tg CO_2 eq.*	*1,000 ha*	*Tg CO_2 eq./yr*	*Tg CO_2 eq./yr*
Eastern Forest Types						
White-red-jack pine	1,252	434	3,423	4,758	(5.7)	1.0
Spruce-fir	1,391	1,163	4,941	6,983	(3.5)	4.1
Longleaf-slash pine	818	396	2,805	5,611	(8.6)	(3.7)
Loblolly-shortleaf pine	3,805	1,742	7,368	21,906	(42.7)	(23.8)
Oak-pine	2,735	1,064	4,156	13,770	(17.1)	(3.3)
Oak-hickory	13,831	3,952	16,877	54,157	(163.4)	(27.4)
Oak-gum-cypress	3,509	1,016	7,087	12,697	(30.5)	(5.6)
Elm-ash-cottonwood	1,232	701	2,482	5,729	(18.1)	(3.8)
Maple-beech-birch	5,998	3,224	11,639	22,748	(43.1)	(1.3)
Aspen-birch	1,307	580	6,370	7,329	(7.6)	(1.5)
Other eastern types	74	20	247	677	26.9	7.5
Nonstocked - East	34	23	219	600	30.4	5.5
Western Forest Types						
Douglas-fir	5,744	3,532	5,269	16,025	49.8	44.1
Ponderosa pine	2,549	1,820	3,291	12,736	58.8	35.9
Western white pine	35	22	37	148	7.5	4.0
Fir-spruce	3,978	2,633	5,532	10,967	32.5	38.1
Hemlock-sitka spruce	1,700	951	2,081	3,613	(0.5)	0.3
Larch	172	106	131	546	(2.6)	(1.0)
Lodgepole pine	1,726	1,039	1,712	7,445	(43.9)	(9.8)
Redwood	215	211	117	373	(1.0)	(0.3)
Hardwoods	2,968	1,763	4,029	13,812	(116.7)	(63.1)
Other western types	1,434	955	1,526	4,618	(112.7)	(40.1)
Pinyon-juniper	2,268	1,808	4,325	20,938	(30.2)	(9.3)
Chaparral	65	57	188	870	16.8	16.8
Nonstocked - West	123	141	652	1,972	(14.3)	(14.8)

Parentheses indicate net sequestration.

Empirical simulation or process models within FORCARB2 estimate ecosystem carbon stocks; estimators are generally classified according to region (Map 4-1), forest type (such as those listed in Table 4-2) and ownership (for example, public versus private forestlands). Live tree carbon and standing dead tree carbon are estimated from stand-level volumes from the inventory data. The volume-to-biomass coefficients are published in Smith and others (2003). Understory carbon is estimated from forest inventory data and equations based on estimates in Birdsey (1996). Forest floor carbon is estimated from the forest inventory data using an empirical simulation model (Smith and Heath 2002). Estimates of carbon in down dead wood are described in Annex O of Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-2001 (EPA 2003a). Estimates of soil carbon are based on data from the STATSGO database (USDA 1991). Soil organic carbon estimates are made solely according to forest type and do not reflect effects of past land use. Some example conversion coefficients used in the forest carbon modeling system are found in Annex O (EPA 2003a).

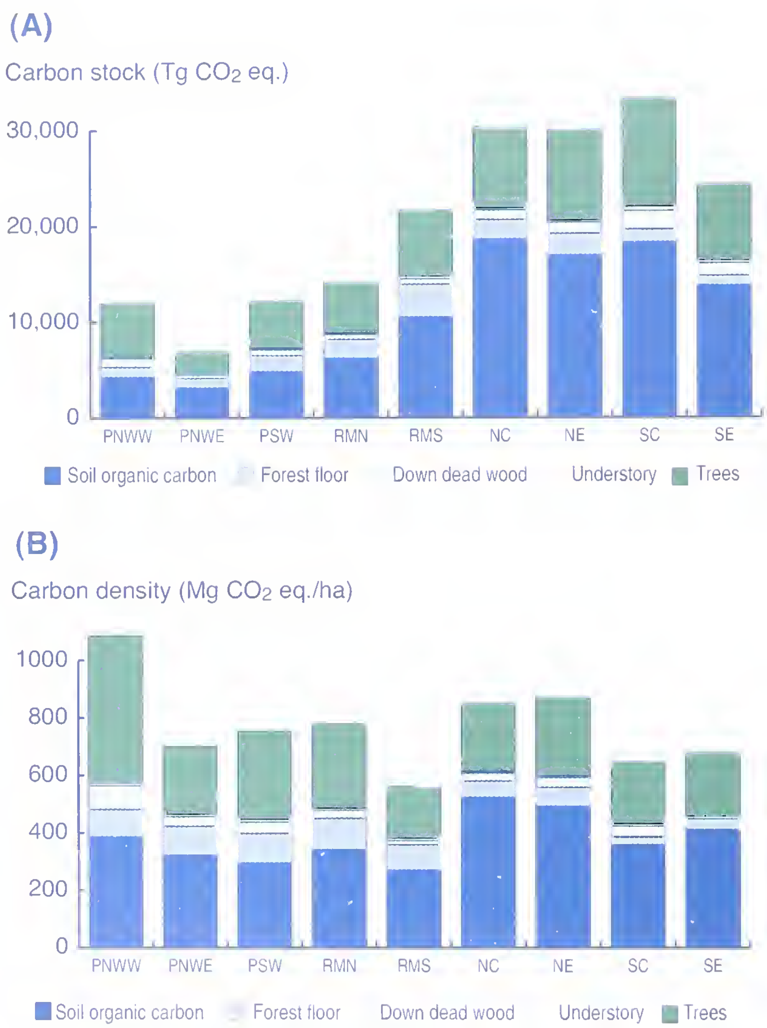
The disposition of harvested wood carbon is simulated according to methods described in Skog and Nicholson (1998). Carbon stocks in wood products in use and wood stored in landfills are based on historical data from the USDA Forest Service (Howard 2001), and historical data as implemented in the framework underlying the NAPAP (Ince 1994) and TAMM/ATLAS (Haynes and others 2003, Mills and Kincaid 1992) models. The carbon conversion factors and decay rates for harvested carbon removed from the forest are taken from Skog and Nicholson (1998). The net carbon stock changes presented in this chapter represent the amounts of carbon that continue to be stored. Annual historical estimates and projections of detailed product production were used to divide consumed roundwood into product, wood mill residue, and pulp mill residue. The rates of carbon decay for products and landfills were estimated and applied to the respective pools. The results were aggregated to produce national estimates. The same disposal rates are used to account for carbon in all wood products produced in the United States, including exported products, whereas carbon in imported wood is not counted.

4.3 Forest Carbon Stocks and Stock Changes, 2001

4.3.1 Estimates by Region, State, Carbon Pool, Forest Type, Age, and Stand Size

Forest areas, carbon stocks, and stock changes for the United States are summarized in Table 4-1. Both carbon dioxide equivalent and carbon summary values are shown for five forest ecosystem pools and two pools of harvested wood products. The rate of increase in ecosystem carbon stocks–net stock change–is somewhat lower since 1997 as compared with the interval from 1987 through 1997. Part of the difference between periods may be due to the timing of forest inventories, or may in part be due to the use of projections after 1997. Organic carbon in soils is the largest carbon pool, followed by carbon in live and standing dead trees. Pools of carbon in harvested wood products are estimated as very similar in size for products in use versus products in landfills. However, the net flux of carbon to landfills is greater. The

Figure 4-2
Forest ecosystem carbon stocks and average stock density according to region and carbon pool, 2001

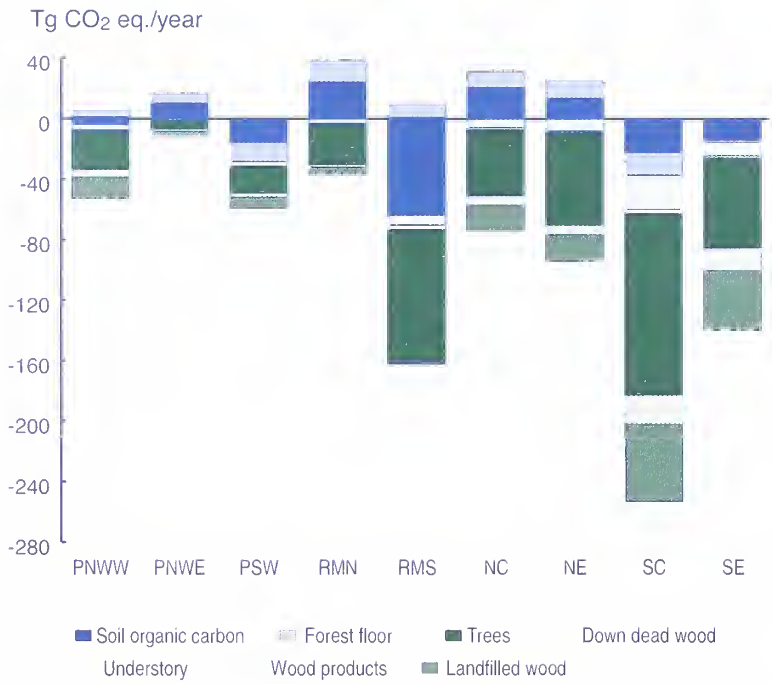


majority of carbon emitted to the atmosphere from harvested wood products is associated with some energy capture from burning.

Total forest ecosystem carbon stocks are greater in the East than in the West (Figure 4-2A). However, regional average values for carbon density, or mass of carbon per unit area (for example, Mg CO_2 eq./ha), do not show such a distinct East-West trend (Figure 4-2B). Thus, the larger pools in the East are principally due to greater forest area (Table 4-2). The most apparent regional trends in ecosystem pool carbon density are: greater carbon in trees in the Pacific Northwest-Westside; greater soil organic carbon pools in northern regions; and smaller pools of down dead wood and forest floor in the South. Net annual stock changes are shown in Figure 4-3, which includes estimated changes in harvested wood product pools. Regional estimates of flux of carbon reemitted to the atmosphere are also included in Figure 4-3. However, the two separate emission fluxes should not be included when summing total flux for a region because those quantities are already included in the net fluxes for wood products and landfills. The greatest net annual increases in carbon stocks were in the South. Area change strongly affects estimates of change in total stocks: regions with a net decrease in forest area since 1997 are the Pacific Northwest-Eastside, Rocky Mountain-North, North Central, Northeast, and Southeast.

Carbon stocks, forest areas, and stock changes for 2001 are shown in Table 4-2 according to forest type. Pool classifications in this table are carbon in biomass, non-living plant mass, and soil. Biomass is live trees plus live understory vegetation. Non-living plant mass includes standing dead trees, down dead wood, and the forest floor. Carbon estimates include aboveground and belowground components. Changes in carbon stock totals as shown in Table 4-2 depend partly on changes in carbon density and partly on changes in total forest area of each forest type. All forest types with a net loss of carbon in biomass during 2001 were accompanied by a net decrease in forest area. Douglas fir, ponderosa pine, and fir-spruce forest types in the West each lost an estimated average of 1 percent of forest area per year since 1997. Other forest types with loss in total carbon stock represent relatively smaller areas of forest, and a portion of area change may be due to reclassification. White-red-jack pine and spruce-fir forests in the East also lost area, but this was estimated at about 1 percent since 1997. For detailed inventory data corresponding to this summary of carbon stocks, see Smith and others (2001).

Figure 4-3
Net annual forest carbon stock change, summarized according to region and carbon pool, 1997-2001



Note: Negative values correspond to sequestration.

Distribution of carbon stocks among forest age classes is shown in Appendix Table C-2 for privately owned and Appendix Table C-3 for publicly owned forests. The tables illustrate that the greater proportion of forest, and thus carbon stocks, in the East are under private ownership while the greater proportion in the West is under public ownership. Distributions according to age are shifted toward older forests on public lands; this is the case for all four regions but is more apparent in the West. Similarly, distribution according to stand size class (Appendix Table C-4) shows a greater proportion in larger size-class stands in the West. Forest land ownership varies by forest type and region. These patterns are illustrated with forest carbon pools (excluding soils) in Appendix Table C-5. Ownership is classified as public or private for timberlands (forests of minimum productivity and available for harvesting). The remaining

forestland, both public and private, is either reserved from harvesting or is considered less productive (and thus not managed for wood products). Western forests include a greater proportion of public and reserve/other forest carbon. Similarly, the greater changes (Appendix Table C-6) are in public and reserve/other forest carbon. For more information about forest inventory variables such as forest classifications of ownership, productivity, forest type, and stand size class see Smith and others (2001), and Miles and others (2001).

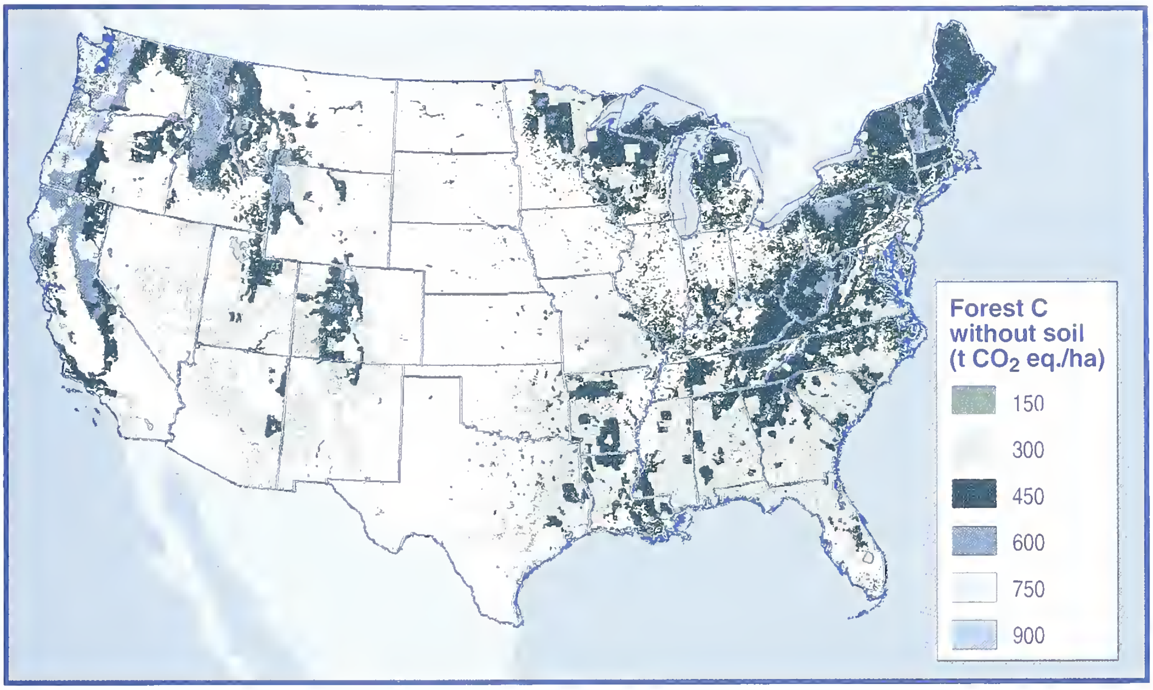
All States in the conterminous United States have forestland. State-by-State totals of forest area, ecosystem carbon stock, stock change, and stock change of harvested wood products for 2001 are shown in Appendix Table C-1. Total stock change per State is the sum of the separate forest ecosystem and products estimates. Estimates of stock change, or flux, for individual States are modeled; a number of States did not have two complete forest inventories as inputs, which can affect the estimates developed by the models. State values for carbon fluxes in wood products, shown in Appendix Table C-1, are based on the amount of wood products produced in the State. Map 4-2 illustrates both the spatial distribution of forest ecosystem carbon and average non-soil carbon density. This map does not include the soil organic carbon and wood products pools, and is therefore not influenced by the relatively large size of this pool. The spatial distributions of net flux of carbon in harvested wood products is influenced by forest management and will not necessarily show the same pattern (these estimates are not yet available in such detail).

4.3.2 Management Activities and Land Use Change

Forest area in the conterminous United States declined by approximately 2.3 million hectares between 1977 and 1987, but area increased by about 4.4 million hectares between 1987 and 1997. Forest area has continued to increase since 1997. These changes in forest area represent average annual fluctuations of less than 0.2 percent. Given the low rate of change in U.S. forestland area, the major influences on the current net carbon flux from forestland are management activities, management intensity, and the ongoing impacts of previous disturbances and past land-use changes. For example, intensified management can increase both the rate of growth and the eventual biomass density of some forest systems, thereby increasing the uptake of carbon. A comparison of carbon density of Southeastern pine stands (10 to 25 years old) found 24 percent greater carbon density in live trees in planted pine stands compared to naturally regenerated stands (156 versus 126 Mg CO_2 eq./ha). Harvesting removes much of the aboveground carbon on forests, but trees may regrow on harvested areas and further sequester carbon. Net ecosystem sequestration associated with harvests depends on factors such as site productivity and decomposition rates. The reversion of cropland to forestland through natural regeneration will cause increased carbon storage in biomass and soils. The net effects of past forest management and land-use change involving forests are captured in the carbon estimates provided in this chapter.

Improved forest management practices, the regeneration of previously cleared forest areas, and timber harvesting and use have resulted in an annual net uptake of carbon during the period

Map 4-2
Average non-soil forest carbon stock density ($t\ CO_2$ eq./ha) over all forestland



from 1987 through 2001 (Table 4-1). Due to improvements in U.S. agricultural productivity, the rate of forest clearing for crop cultivation and pasture slowed in the late 19th century, and by 1920 this practice had all but ceased. As farming expanded in the Midwest and West, large areas of previously cultivated land in the East were taken out of crop production, primarily between 1920 and 1950, and were allowed to revert to forests or were actively reforested. The impacts of these land-use changes are still affecting carbon fluxes for forests in the East. In addition to land-use changes in the early part of this century, carbon fluxes from Eastern forests have been affected by a trend toward managed growth on private land. Collectively, these changes have produced a near doubling of the carbon density in Eastern forests since the early 1950s. More recently, the 1970s and 1980s saw a resurgence of federally supported forest management programs (for example, the Forestry Incentive Program) and soil conservation programs (for example, the Conservation Reserve Program), which have focused on tree planting, improving timber management activities, combating soil erosion, and converting marginal cropland to forests. In addition to forest regeneration and management, forest harvests have also affected net carbon fluxes. Because most of the timber that is harvested from U.S. forests is used in wood products and much of the discarded wood products are disposed of in landfills, rather than by incineration, significant quantities of this harvested carbon are transferred to long-term storage pools rather than being released to the atmosphere. The size of these long-term carbon storage pools has increased over the last century.

A large proportion of non-forest trees in the United States are in urban areas. These urban trees constitute approximately 3 percent of total tree cover in the conterminous United States (EPA 2003a). Methods have been developed for estimating carbon sequestration rates for urban trees of the United States (Nowak and Crane 2002). Net flux of carbon into urban trees for 2001 was estimated at 58.7 Tg CO_2 eq. per year (from Table 6-6 in EPA 2003a). This represents significant carbon accumulation (Table 4-1). Urban trees provide additional favorable GHG benefits beyond accumulating carbon stocks; these include direct effects on energy savings [see Dwyer and others (2000) and Akbari (2002) for further discussion].

4.4 Uncertainty of the Estimates

Carbon stocks and stock changes as provided here are the current estimates of the most likely values, and include some level of uncertainty. Better information such as samples, processes, or models could reduce uncertainty of future estimates. However, we believe the estimates are unbiased, so reductions in uncertainty are not expected to change mean values significantly.

Forest inventories, which are the input data to FORCARB2, include sampling, measurement and modeling errors that contribute to uncertainty. Forest Inventory and Analysis surveys are based on a statistical sample designed to represent the wide variety of growth conditions present over large territories. Although the potential for uncertainty is large, the sample design for forest surveys contributes to limiting the error, and relative error is inversely proportional to degree of aggregation of inventory data (Phillips and others 2000). Similarly, the inventory design contributes to limiting uncertainty about net annual carbon stock change. Estimates from sampling at different times on permanent plots are correlated, and this correlation reduces the uncertainty in stock change variables (Smith and Heath 2001).

Additional sources of uncertainty come from the empirical models used by FORCARB2 to estimate carbon storage in specific ecosystem components, such as forest floor, understory vegetation, and mineral soil. Certainty about model predictions is limited by precision in process definitions, coefficients, and relationships among system components. Uncertainty also arises from extrapolation of the results of site-specific ecosystem studies to very large areas of forestland because such studies may not adequately represent regional or national averages. An important source of uncertainty is attributed to the lack of knowledge about the impacts of forest management activities, including harvest, on soil carbon. Soil carbon impact estimates need to be very precise because even small changes in soil carbon may sum to large differences over large areas; thus, limited understanding of soils can significantly affect overall forest carbon budget estimates (Heath and others 2003).

4.5 Summary of Current Net CO_2 Sequestration for U.S. Forests and Forest Products

Forest ecosystems and forest products represent a significant carbon dioxide sink in the United States. Over 90 percent of the sequestration in agriculture and forests occurs in the forest

sector, with an additional 7 percent sequestered in urban trees. Total carbon stocks in forest ecosystems of the conterminous United States are estimated at 184,800 Tg CO_2 eq. (Table 4-1). The net amount of carbon stored in forest ecosystems in the conterminous U.S. increased by an estimated 547 Tg CO_2 eq. This estimate does not include increases in biomass harvested from a portion of U.S. forests, used largely as timber and fuelwood. In the same year, the net increase in carbon sequestered in harvested wood products, which includes long-term storage in landfills, is estimated at 212 Tg CO_2 eq. This net value is the sum of an annual sink in harvested wood of 533 Tg CO_2 eq. and emission of carbon to the atmosphere of 321 Tg CO_2 eq. Fifty eight percent of these emissions included some form of energy recapture associated with the combustion of wood products. Total net sequestration, or gain in carbon storage, by forest ecosystems and harvested wood products for 2001 was 759 Tg CO_2 eq.

5.1 Sources of Greenhouse Gas Emissions from Energy Use on Agricultural Operations

Agriculture operations, such as crop and livestock farms, dairies, nurseries and greenhouses, use energy from a variety of sources. Energy use depends in part on the size and sort of agricultural operation and trends in fuel costs. While energy use in agriculture causes CO_2 emissions, this source is small relative to the total U.S. emissions of CO_2 from energy.

Energy is used directly in agriculture for a range of purposes, including operating vehicles and irrigation pumps, and controlling indoor temperatures of greenhouses, barns, and other farm buildings. Crop production requires a large amount of liquid fuel for field operations. Most large farms use diesel-fueled vehicles for tilling, planting, cultivating, disking, harvesting, and applying chemicals. Gasoline is used for small trucks and older harvesting equipment primarily. Smaller farms are more likely to use gasoline-powered equipment, but as farms get larger they tend to use more diesel fuel. In addition, energy is used in some operations to dry crops such as grain, tobacco, and peanuts; and livestock operations use energy to operate various types of equipment. Indirectly, GHG emissions result from energy consumption and other processes in manufacturing of agricultural inputs such as fertilizer, lime, and other soil amendments. These indirect emissions are not detailed in this inventory. The U.S. GHG Inventory addresses energy and non-energy processes in industry and manufacturing by general end-use sectors rather than specific end uses such as agriculture.

While many irrigation systems in the United States are gravity flow systems that require little or no energy for water distribution, irrigation systems that use pumps to distribute water use energy. Based on the most recent USDA Farm and Ranch Irrigation Survey, in 1998 about 38 million acres of U.S. farmland were irrigated with pumps powered by liquid fuels, natural gas, and electricity, costing a total of $1.2 billion ($32 per irrigated acre) (USDA NASS 1999b). Electricity was the main power source for these pumps, costing $801 million for 20 million acres. Diesel fuel was used to power pumps on about 10 million acres and natural gas was used on about 6 million acres.

The area of land irrigated can vary substantially from year to year, depending on environmental conditions. For example, in 1998, 50 million acres of farmland in the United States were irrigated (including gravity flow irrigation), about 4 million acres more than were irrigated in 1994 (USDA NASS 1999b). Corn for grain or seed, alfalfa hay, cotton, soybeans, and orchard land (e.g., fruit trees, vineyards, and nut trees) required the most water in 1998, accounting for 57% of all irrigated land. California irrigated the most land, covering about 8.1 million acres of farmland in 1998. Other leading irrigation States include Nebraska, Texas, and Arkansas.

The amount and type of energy used in agriculture operations affect CO_2 emissions. Generally CO_2 levels increase with higher energy use. Some fuels have higher carbon content than others, resulting in higher CO_2 emissions per Btu burned (Table 5-1). However, some fuel/engine applications are more energy efficient than others and require less fuel to perform similar operations. For example, diesel fuel has a higher Btu content than natural gas on a volumetric

basis and diesel engines have a higher performance rating compared to engines designed to run on natural gas. Therefore, even though diesel fuel has higher carbon content per Btu compared to natural gas, using diesel engines to perform some farm operations may result in lower CO_2 emissions.

Table 5-1 Energy use and carbon dioxide emissions by fuel source on U.S. farms, 2001

Fuels	Energy consumed		Carbon content	Fraction oxidized	CO_2 emissions
	Trillion Btu	*Qbtu*	*Tg C/ Qbtu*		*Tg CO_2 eq.*
Diesel	484	0.484	19.95	0.99	35.04
Gasoline	144	0.144	19.34	0.99	10.11
LP gas	66	0.066	16.99	0.99	4.07
Natural gas	58	0.058	14.47	0.995	3.06
Electricity	370	0.37	**	**	58.81
Total	1,122	1.122			111.09

** Varies depending on fuel used to generate electricity and heat rate of the power generating facility.

Electricity is generated from multiple fuels, the proportions of which may vary from year to year, affecting overall emissions. It is assumed that electricity used on agricultural operations is generated from coal, natural gas, or petroleum. Emission factors are higher for fuels when they are used to generate electricity because there is a heat loss associated with various power generating facilities.

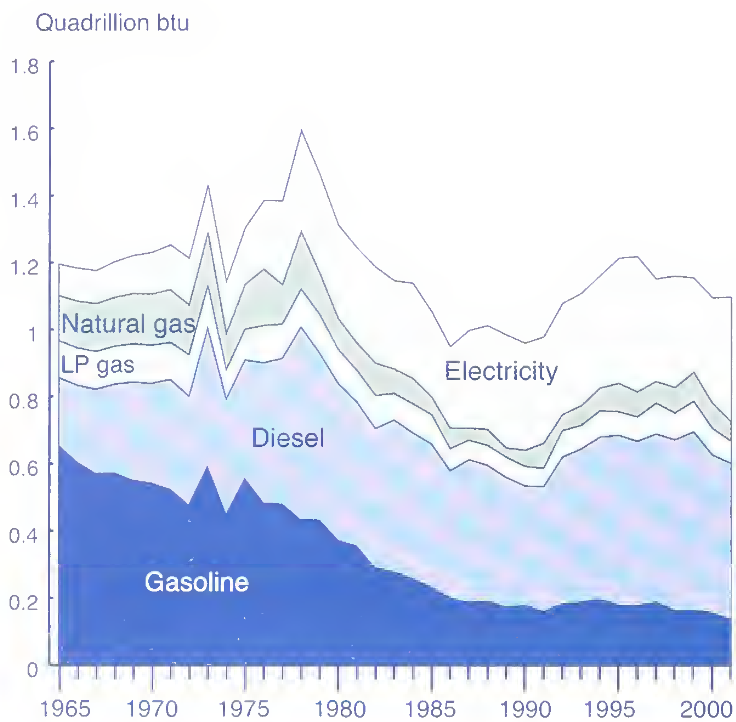
5.2 Summary of Greenhouse Gas Emissions from Energy Use in Agriculture

Over 1 Qbtu of energy was used directly in agriculture in 2001, resulting in about 111 Tg CO_2 emissions (Table 5-1). The same year, total energy consumption for all sectors in the United States, including agriculture, was 82 Qbtu, resulting in emissions of 5,597 Tg CO_2. Agriculture's contribution to this total was very small at about 2 percent. Within agriculture, electricity accounted for about 53 percent and diesel fuel for about 31 percent of CO_2 emissions from energy use (Table 5-1). Gasoline consumption accounted for about 9 percent of CO_2 emissions, while liquefied petroleum (LP) gas and natural gas accounted for about 4 percent, and 3 percent respectively.

Emissions from fuel consumption vary regionally across the United States. The highest emissions are in the Northern Plains and Corn Belt States (Table 5-2). Intermediate emissions occur in the Pacific, Southern Plains, Mountain, and Lake States. Relatively small emissions are estimated for the Southeast, Northeast, Delta, and Appalachian States. There is a strong correlation between production and energy use/emissions. The States with the most agricultural production use the most energy.

5.3 Long-term Trends in Greenhouse Gas Emissions from Energy Use in Agriculture

Figure 5-1
Energy use in agriculture by source, 1965-2001



Agricultural energy use and resulting CO_2 emissions grew throughout the 1960s and 1970s, peaking in the late 1970s (Figure 5-1). High prices, stemming from the oil crisis of the late 1970s and early 1980s, led farmers to be more energy-efficient, driving a decline in energy use and CO_2 emissions throughout most of the 1980s. This decline is attributed to switching from gasoline-powered to more fuel-efficient diesel-powered engines, adopting energy-conserving tillage practices, shifting to larger multifunction machines, and adopting energy-saving methods of crop drying and irrigation (Uri and Day 1991; USDA ERS 1994). Farm energy use leveled off in the late 1980s as energy prices subsided (Figure 5-1). Since 1990 there has been a small upward trend in energy use; however, farm energy used today is still well below the peak levels of the 1970s. Moreover, energy productivity, i.e., output per unit of energy input, has increased significantly.

One of the most notable changes in farm energy consumption over the past 30 years is the substitution of diesel fuel for gasoline (Figure 5-2). Gasoline use dropped from 41 percent of total energy used on farms in 1965 to only 8 percent in 2001, while diesel's share of total energy rose from 13 to 28 percent. Producers switched to diesel fuel equipment as farms grew in size—average farm size in 1965 was 340 acres compared to 434 acres in 2000 (USDA NASS 2002). As farmers scaled up their operations they began to purchase large-scale equipment with more horsepower. Heavy-duty vehicles generally are powered by diesel engines because they are more energy efficient than gasoline engines (Uri and Day 1991). Diesel-powered equipment has become the standard on U.S. farms.

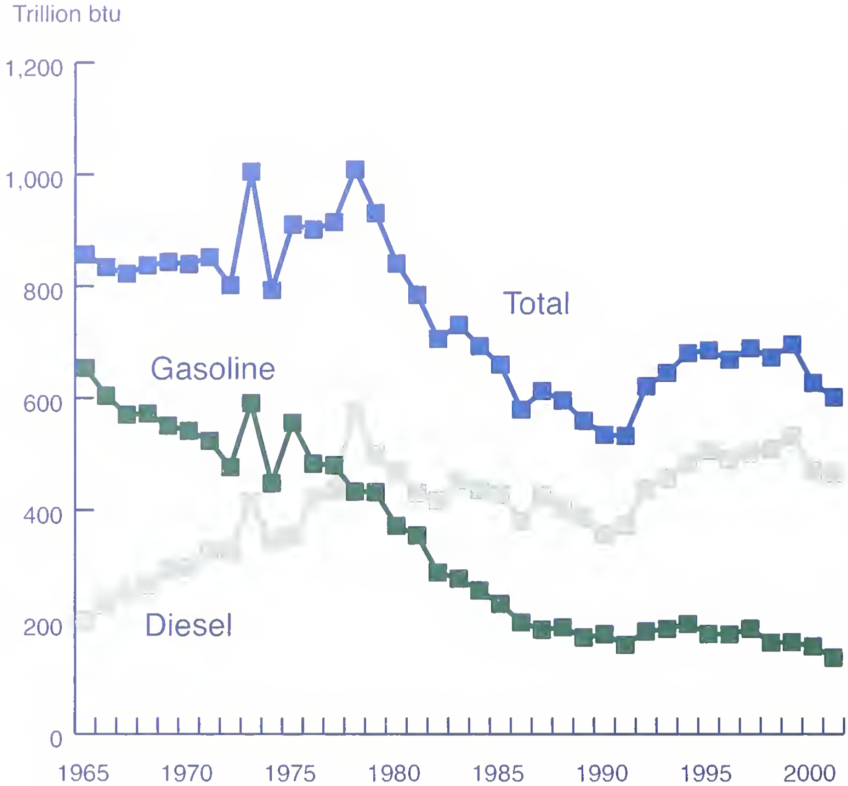
Another major change in farm energy consumption began around 1979 when automobile manufacturers began producing more fuel-efficient vehicles. Laws such as the Energy Policy and Conservation Act of 1975 increased average fuel economy standards and both gasoline- and diesel-powered equipment became increasingly energy efficient throughout the 1980s and

Table 5-2 CO_2 emissions from energy use in agriculture, by region, 2001

Region	States	Energy Emissions
		Tg CO_2 eq.
Northeast	Maine	5.4
	New Hampshire	
	Vermont	
	Massachusetts	
	Rhode Island	
	Connecticut	
	New York	
	New Jersey	
	Pennsylvania	
	Delaware	
	Maryland	
	District of Columbia	
Lake States	Michigan	12.9
	Wisconsin	
	Minnesota	
Corn Belt	Ohio	18.5
	Indiana	
	Illinois	
	Iowa	
	Missouri	
Northern Plains	North Dakota	17.6
	South Dakota	
	Nebraska	
	Kansas	
Appalachian	Virginia	8.2
	West Virginia	
	North Carolina	
	Kentucky	
	Tennessee	
Southeast	South Carolina	6.2
	Georgia	
	Florida	
	Alabama	
Delta States	Mississippi	6.3
	Arkansas	
	Louisiana	
Southern Plains	Oklahoma	12.2
	Texas	
Mountain	Montana	11.6
	Idaho	
	Wyoming	
	Colorado	
	New Mexico	
	Arizona	
	Utah	
	Nevada	
Pacific	Washington	12.3
	Oregon	
	California	

Note: Does not include Alaska and Hawaii.

Figure 5-2
Gasoline and diesel fuel used on farms, 1965-2001



1990s. This is reflected in Figure 5-2, which shows total fuel consumption peaking in 1978 at about 1 Qbtu and falling to a low of 531 trillion btu in 1991.

The adoption of energy-conservation tillage practices has contributed to decreasing fuel use on farms in the United States. Conservation tillage leaves roughly 30 percent of plant residues on the soil surface after planting. It requires far less energy than conventional-till, which involves extensive field preparation prior to planting, and removes 70 percent or more of the plant residue from the soil (Lin et al. 1995). In the most extreme case, using a moldboard plow for conventional-till, almost 100 percent of the plant residue is removed. Adoption of conservation-till on major field crops, such as corn and soybeans, began to increase significantly in the 1980s. For example, on corn acreage between 1987 and 1992, the use of the moldboard plow declined from 21 to 12 percent and conventional tillage without the moldboard plow dropped from 60 to 49 percent (Lin et al. 1995). Conservation tillage methods, such as mulch-till, increased from 14 to 25 percent, and no-ridge-till rose from 5 to 14 percent.

5.4 Methods for Estimating CO_2 Emissions from Energy Use in Agriculture

CO_2 emission estimates for energy use are constructed from fuel consumption data using standardized methods published in the U.S. GHG Inventory (EPA 2003a). Emission estimates from fuel use in agriculture are not explicitly published in the U.S. GHG Inventory; however, they are contained in the estimates of fuel consumption and emissions by sectors. The emissions estimates presented in this chapter were prepared separately from the U.S. GHG Inventory.

Estimates of CO_2 emissions from agricultural operations are based on energy data from the Agricultural Resource Management Survey (ARMS) conducted by the National Agricultural

Statistics Service (NASS) of the USDA. The ARMS collects information on farm production expenditures including expenditures on diesel fuel, gasoline, LP gas, natural gas, and electricity (USDA NASS, 2001b). NASS also collects data on prices paid by farmers for gasoline, diesel, and LP gas (USDA NASS 2001b). Energy expenditures and fuel prices are used to approximate consumption by dividing expenditures by purchase prices. NASS aggregates individual State data into 10 production regions, allowing for fuel consumption to be estimated at the national and regional levels. Electricity and natural gas prices, which are not collected by NASS, are from the Energy Information Administration (DOE EIA) of the Department of Energy that reports average electricity and natural gas prices by State (DOE EIA 2001). State-level price data were aggregated by NASS region to produce estimates of regional prices.

Table 5-3 Average emission factors for 1998-2000 by utility and non-utility generators by USDA NASS regions

Region	tons CO_2/MWh
Appalachian	0.767
Corn Belt	0.843
Delta States	0.616
Lake States	0.792
Mountain	0.781
Northeast	0.534
Northern Plains	0.843
Pacific	0.224
Southeast	0.636
Southern Plains	0.749

Following the method outlined in Annex A of the U.S. GHG Inventory (EPA 2003a), consumption of diesel fuel, gasoline, LP gas, and natural gas was converted to CO_2 emissions using coefficients for carbon content of fuels and fraction of carbon oxidized during combustion, both of which are published in Annex A of EPA (2003a) and provided in Table 5-1. These carbon content coefficients were derived by EIA and are similar to those published by the Intergovernmental Panel on Climate Change (IPCC). For each fuel type, fuel consumption in units of Qbtu was multiplied by the carbon content coefficient (Tg C/Qbtu) to estimate Tg of carbon contained in the fuel consumed. This value is sometimes referred to as "potential emissions" because it represents the maximum amount of carbon that could be released to the atmosphere if all carbon were oxidized (EPA 2003a). However, only a portion of the carbon is actually oxidized during combustion. Table A-15 in Annex A of the U.S. GHG Inventory provides coefficients for the fraction of carbon oxidized during combustion by fuel type (EPA 2003a). These coefficients are also shown in Table 5-1 of this report. It is assumed that 100 percent of the carbon oxidized is emitted to the atmosphere as CO_2. CO_2 emissions are estimated by multiplying potential emissions by the fraction of carbon oxidized for each fuel type.

A different approach was used to estimate emissions from electricity, since a number of fuel sources can be used to generate electricity. Also, fuel sources vary significantly by region; for example, some regions of the country rely more on coal for electricity generation, while other regions use more natural gas to generate electricity. The mix of fuel sources used in a region

can change from year to year. To address this variation, CO_2 emissions from electricity generation were derived from adjusted emission factors obtained from EIA. EIA typically reports CO_2 emissions from electricity generation by State and U.S. Census Region (DOE EIA 2002). In response to a special request from USDA, EIA aggregated their State emission factors into the NASS production regions. The regional-level electricity emission factors represent average CO_2 emissions generated by utility and non-utility electric generators for the 1998-2000 time period (Table 5-3). These regional emission factors were multiplied by estimated electricity use in each NASS farm production region to calculate CO_2 emissions.

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Appendix A: Livestock Emissions

Appendix Table A-1 Population of animals in each State, 2001

	Beef cattle	Dairy cattle	Swine	Sheep	Goat	Horse	Poultry[1]
				Head			
Alabama	1,459,094	31,452	195,000	9,867	27,578	93,169	199,103,000
Alaska	9,752	1,475	1,000	9,867	167	ND	0
Arizona	776,920	169,005	133,000	132,000	41,000	82,694	0
Arkansas	1,909,130	51,089	633,750	9,867	17,083	87,771	245,163,697
California	3,215,926	2,264,980	110,000	840,000	40,191	247,722	34,104,205
Colorado	2,987,013	132,305	775,000	420,000	12,841	178,855	4,210,000
Connecticut	29,920	36,806	3,500	8,333	1,460	14,886	3,749,629
Delaware	16,244	12,287	26,000	9,867	674	7,113	48,187,848
Florida	1,751,208	192,471	35,000	9,867	24,209	120,188	34,086,636
Georgia	1,286,886	115,101	310,000	9,867	36,637	77,354	257,954,818
Hawaii	152,856	9,819	27,000	9,867	3,305	ND	822,636
Idaho	1,493,988	509,077	24,000	275,000	6,269	130,440	1,225,000
Illinois	1,364,354	174,411	4,150,000	75,000	10,781	113,206	5,080,625
Indiana	730,055	209,239	3,162,500	66,000	11,664	128,401	33,518,261
Iowa	3,409,002	323,140	14,950,000	270,000	12,275	132,328	44,157,704
Kansas	6,551,039	161,886	1,552,500	110,000	7,293	115,699	2,139,000
Kentucky	2,267,812	167,516	405,000	9,867	13,877	210,101	51,672,727
Louisiana	854,078	68,223	26,000	9,867	8,315	65,975	2,700,000
Maine	46,323	56,334	6,500	8,333	2,214	12,593	5,607,000
Maryland	134,907	113,939	52,000	9,867	5,311	49,350	56,791,110
Massachusetts	21,032	29,456	18,000	8,333	2,490	20,609	373,104
Michigan	606,015	422,144	937,500	71,000	10,603	144,987	9,470,798
Minnesota	1,913,768	792,741	5,825,000	170,000	7,871	122,357	38,117,066
Mississippi	1,108,265	50,091	285,000	9,867	18,389	69,188	149,533,455
Missouri	4,357,024	211,072	3,000,000	73,000	22,683	187,670	16,279,590
Montana	2,500,403	28,403	170,000	360,000	4,909	155,920	480,000
Nebraska	6,390,010	103,176	2,912,500	114,000	5,185	100,390	14,322,182
Nevada	507,643	36,279	7,000	95,000	1,828	31,143	0
New Hampshire	19,074	25,517	3,500	8,333	2,368	10,219	216,954
New Jersey	29,139	21,634	13,000	9,867	4,040	49,534	2,360,332
New Mexico	1,347,289	312,437	3,000	255,000	45,294	85,011	0
New York	501,358	947,192	75,000	60,000	14,322	104,588	5,314,306
North Carolina	946,349	97,187	9,675,000	9,867	36,892	90,182	161,636,606
North Dakota	1,915,644	59,129	154,000	138,000	6,315	76,879	521,173
Ohio	947,591	373,051	1,452,500	142,000	20,003	166,993	46,863,427
Oklahoma	5,269,788	108,714	2,320,000	55,000	31,967	205,239	46,631,364
Oregon	1,246,188	146,461	29,000	245,000	16,608	149,531	3,704,000
Pennsylvania	873,198	873,144	1,050,000	81,000	20,403	142,514	56,477,008
Rhode Island	3,572	2,635	2,500	8,333	298	2,446	0
South Carolina	453,531	31,398	320,000	9,867	27,153	50,331	45,942,743
South Dakota	3,942,645	129,130	1,277,500	420,000	5,101	113,393	4,373,091
Tennessee	2,249,948	137,296	225,000	9,867	49,755	194,956	38,304,545
Texas	14,022,015	438,754	887,500	1,150,000	1,280,431	529,963	128,919,182
Utah	802,534	138,240	610,000	390,000	5,974	107,506	3,775,000
Vermont	98,209	213,232	2,500	8,333	2,617	19,426	221,658
Virginia	1,602,670	172,578	410,000	61,000	19,830	110,125	61,735,226
Washington	866,879	343,704	24,000	54,000	9,509	128,708	6,372,000
West Virginia	400,288	21,688	11,000	35,000	7,572	36,765	19,615,778
Wisconsin	1,633,252	1,921,876	565,000	80,000	20,071	114,866	11,790,909
Wyoming	1,575,754	5,935	117,000	530,000	6,174	110,716	17,000
Total	88,597,582	12,994,849	58,959,750	6,965,000	1,989,799	5,300,000	2,054,998,477

Source: USDA NASS (2002):cattle, cattle on feed, chicken and eggs, hogs and pigs, sheep and goats; FAO (2002) for horses.

[1] 151,356,084 head from unspecified States are included in the total, but not represented in State totals for confidentiality.

Appendix Table A-2 U.S. livestock population, 1990-2001

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
						1,000 head						
Dairy cattle	14,143	13,980	13,830	13,767	13,566	13,502	13,305	13,138	12,992	13,026	13,070	12,995
Dairy cows	10,007	9,883	9,714	9,679	9,504	9,491	9,410	9,309	9,200	9,142	9,220	9,166
Dairy heifers	4,135	4,097	4,116	4,088	4,062	4,011	3,895	3,829	3,793	3,884	3,850	3,828
Swine	53,941	56,478	58,532	58,016	59,951	58,899	56,220	58,728	61,991	60,245	58,892	58,960
Market swine	47,043	49,247	51,276	50,859	52,669	51,973	49,581	51,888	55,150	53,871	52,658	52,772
Market <60 lbs.	18,359	19,212	19,851	19,434	20,157	19,656	18,851	19,886	20,691	19,928	19,582	19,628
Market 60-119 lbs.	11,734	12,374	12,839	12,656	13,017	12,836	12,157	12,754	13,552	13,256	12,933	12,940
Market 120-179 lbs.	9,440	9,840	10,253	10,334	10,671	10,545	10,110	10,480	11,235	11,043	10,753	10,747
Market >180 lbs.	7,510	7,822	8,333	8,435	8,824	8,937	8,463	8,768	9,672	9,645	9,390	9,457
Breeding swine	6,899	7,231	7,255	7,157	7,282	6,926	6,639	6,840	6,841	6,374	6,233	6,188
Beef cattle	86,087	87,267	88,548	90,321	92,571	94,391	94,269	92,290	90,730	90,032	89,215	88,598
Feedlot steers	7,338	7,920	7,581	7,984	7,797	7,763	7,380	7,644	7,845	7,781	8,280	8,566
Feedlot heifers	3,621	4,035	3,626	3,971	3,965	4,047	3,999	4,396	4,459	4,578	4,872	5,035
Bulls not on feed	2,180	2,198	2,220	2,239	2,306	2,392	2,392	2,325	2,235	2,241	2,196	2,187
Calves not on feed	23,909	23,853	24,118	24,209	24,586	25,170	25,042	24,363	24,001	23,895	23,508	22,953
Heifers not on feed	8,872	8,938	9,520	9,850	10,469	10,680	10,869	10,481	9,998	9,725	9,353	9,224
Steers not on feed	7,490	7,364	8,031	7,935	8,346	8,693	9,077	8,452	8,050	7,864	7,248	7,009
Cows not on feed	32,677	32,960	33,453	34,132	35,101	35,645	35,509	34,629	34,143	33,948	33,760	33,624
Sheep	11,358	11,174	10,797	10,201	9,836	8,989	8,465	8,024	7,825	7,215	7,032	6,965
Sheep not on feed	10,271	10,168	9,748	9,151	8,940	8,193	7,697	7,270	7,085	6,519	6,351	6,290
Sheep on feed	1,088	1,055	1,049	1,050	896	797	768	754	740	696	681	675
Goats	2,516	2,516	2,516	2,410	2,305	2,200	2,095	1,990	1,990	1,990	1,990	1,990
Poultry	1,537,074	1,594,944	1,649,998	1,707,422	1,769,135	1,679,704	1,882,078	1,926,790	1,963,919	2,007,284	2,072,877	2,054,998
Hens	119,551	117,178	121,103	131,688	135,094	133,841	138,048	140,966	150,778	151,914	153,232	153,237
Pullets	227,083	239,559	243,267	240,712	243,286	246,599	247,446	261,515	265,634	274,520	273,801	279,726
Chickens	6,545	6,857	7,113	7,240	7,369	7,637	7,243	7,549	7,682	9,659	8,088	8,126
Broilers	1,066,209	1,115,845	1,164,089	1,217,147	1,275,916	1,184,667	1,381,229	1,411,673	1,442,596	1,481,093	1,549,818	1,525,291
Turkeys	117,685	115,504	114,426	110,635	107,469	106,960	108,112	105,088	97,229	90,098	87,938	88,619
Horses	5,069	5,100	5,121	5,130	5,110	5,130	5,150	5,170	5,237	5,170	5,240	5,300

Source: USDA NASS (2002): cattle, cattle on feed, chicken and eggs, hogs and pigs, and sheep and goats; FAO (2002) for horses.

Appendix Table A-3 State-level methane emissions from enteric fermentation, 1990-2001

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
						Tg CO_2 eq.						
Alabama	1.79	1.74	1.77	1.78	1.84	1.94	1.91	1.74	1.70	1.64	1.62	1.51
Alaska	0.01	0.01	0.01	0.01	0.01	0.01	0.02	0.02	0.02	0.02	0.01	0.02
Arizona	1.10	1.10	1.15	1.14	1.14	1.12	1.11	1.07	1.11	1.11	1.08	1.08
Arkansas	2.05	1.95	1.97	1.95	2.12	2.24	2.17	2.15	2.06	2.05	2.07	2.07
California	6.61	6.42	6.51	6.17	6.44	6.50	6.06	6.21	6.20	6.50	6.59	6.72
Colorado	3.15	2.99	3.26	3.18	3.20	3.22	3.25	3.30	3.32	3.26	3.18	3.12
Connecticut	0.12	0.12	0.12	0.12	0.12	0.12	0.11	0.11	0.11	0.11	0.11	0.10
Delaware	0.04	0.04	0.05	0.05	0.04	0.04	0.04	0.04	0.04	0.04	0.04	0.04
Florida	2.52	2.53	2.54	2.52	2.59	2.64	2.50	2.45	2.34	2.28	2.29	2.26
Georgia	1.64	1.66	1.67	1.65	1.69	1.72	1.65	1.60	1.50	1.47	1.48	1.44
Hawaii	0.25	0.27	0.24	0.23	0.22	0.23	0.22	0.23	0.24	0.23	0.22	0.20
Idaho	2.26	2.30	2.38	2.33	2.37	2.51	2.52	2.56	2.63	2.67	2.77	2.83
Illinois	2.12	2.09	2.14	2.14	2.08	2.00	1.87	1.82	1.80	1.82	1.80	1.74
Indiana	1.50	1.44	1.42	1.37	1.42	1.42	1.34	1.34	1.27	1.28	1.23	1.16
Iowa	5.05	4.95	5.01	4.79	4.67	4.75	4.43	4.32	4.21	4.34	4.27	4.15
Kansas	5.70	5.56	5.55	5.62	5.70	6.01	5.95	5.94	5.82	5.97	5.94	5.98
Kentucky	2.89	2.94	3.01	2.97	2.99	3.03	3.00	2.90	2.80	2.69	2.52	2.57
Louisiana	1.34	1.28	1.21	1.24	1.20	1.22	1.20	1.19	1.17	1.14	1.14	1.10
Maine	0.17	0.17	0.17	0.17	0.17	0.16	0.16	0.16	0.16	0.16	0.15	0.15
Maryland	0.47	0.47	0.45	0.46	0.45	0.45	0.42	0.39	0.38	0.38	0.37	0.35
Massachusetts	0.13	0.12	0.13	0.12	0.12	0.11	0.11	0.10	0.10	0.10	0.10	0.08
Michigan	1.52	1.47	1.49	1.50	1.52	1.52	1.46	1.41	1.37	1.37	1.34	1.31
Minnesota	3.37	3.42	3.44	3.45	3.25	3.38	3.34	3.30	3.18	3.15	3.20	3.16
Mississippi	1.51	1.50	1.52	1.57	1.56	1.51	1.53	1.43	1.36	1.31	1.24	1.25
Missouri	5.08	5.10	5.14	5.24	5.46	5.42	5.39	5.32	5.11	5.10	4.98	4.94
Montana	3.11	3.25	3.49	3.44	3.43	3.63	3.60	3.56	3.49	3.49	3.53	3.48
Nebraska	5.74	5.84	5.91	5.82	6.10	6.21	6.43	6.49	6.62	6.69	6.58	6.39
Nevada	0.69	0.69	0.72	0.70	0.69	0.71	0.68	0.71	0.70	0.70	0.72	0.73
New Hampshire	0.07	0.08	0.08	0.08	0.08	0.08	0.07	0.07	0.07	0.08	0.08	0.07
New Jersey	0.13	0.13	0.13	0.12	0.12	0.11	0.11	0.11	0.10	0.10	0.09	0.09
New Mexico	1.91	1.88	1.92	1.99	2.03	2.12	2.22	2.24	2.30	2.33	2.35	2.35
New York	2.49	2.50	2.54	2.43	2.40	2.37	2.27	2.29	2.32	2.34	2.35	2.26
North Carolina	1.15	1.22	1.29	1.31	1.39	1.49	1.53	1.52	1.39	1.40	1.38	1.39
North Dakota	2.19	2.25	2.28	2.34	2.46	2.53	2.42	2.41	2.29	2.39	2.40	2.55
Ohio	1.96	1.95	1.92	1.81	1.78	1.79	1.79	1.70	1.67	1.60	1.59	1.59
Oklahoma	5.49	5.53	5.59	5.50	5.40	5.83	5.70	5.57	5.63	5.49	5.48	5.40
Oregon	2.02	2.02	2.01	1.99	2.11	2.21	2.24	2.22	2.17	2.15	2.11	2.01
Pennsylvania	2.67	2.66	2.74	2.60	2.53	2.53	2.45	2.43	2.42	2.40	2.39	2.37
Rhode Island	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01	0.01
South Carolina	0.67	0.59	0.62	0.63	0.61	0.61	0.61	0.59	0.56	0.56	0.55	0.53
South Dakota	4.20	4.18	4.35	4.62	4.49	4.80	4.59	4.47	4.32	4.52	4.62	4.91
Tennessee	2.61	2.59	2.59	2.58	2.72	2.80	2.76	2.56	2.48	2.38	2.35	2.39
Texas	14.78	14.53	15.27	15.53	16.22	16.47	16.11	15.16	15.12	15.03	14.59	14.33
Utah	1.23	1.24	1.23	1.30	1.30	1.34	1.36	1.39	1.39	1.35	1.40	1.42
Vermont	0.52	0.51	0.51	0.51	0.49	0.50	0.48	0.48	0.50	0.51	0.50	0.49
Virginia	1.97	1.98	2.00	1.94	1.93	1.96	1.95	1.95	1.86	1.84	1.76	1.81
Washington	1.96	1.99	1.98	2.00	2.02	1.94	1.85	1.79	1.76	1.73	1.77	1.70
West Virginia	0.57	0.59	0.60	0.59	0.58	0.59	0.57	0.53	0.52	0.51	0.49	0.47
Wisconsin	5.64	5.59	5.51	5.36	5.13	5.11	4.98	4.88	4.85	4.79	4.83	4.77
Wyoming	1.66	1.66	1.78	1.86	2.00	1.95	1.94	2.09	2.13	2.01	2.03	1.98
Total	117.85	117.10	119.39	118.82	120.40	122.96	120.47	118.33	116.70	116.58	115.68	114.82

Appendix Table A-4 State-level methane emissions from enteric fermentation by livestock category, 2001

	Beef cattle	Dairy cattle	Swine	Sheep	Goats	Horses
			Tg CO_2 eq.			
Alabama	1.40	0.06	0.01	0.00	0.00	0.04
Alaska	0.01	0.00	0.00	0.00	0.00	**
Arizona	0.59	0.43	0.00	0.02	0.00	0.03
Arkansas	1.91	0.10	0.02	0.00	0.00	0.03
California	2.11	4.37	0.00	0.14	0.00	0.09
Colorado	2.69	0.27	0.02	0.07	0.00	0.07
Connecticut	0.02	0.08	0.00	0.00	0.00	0.01
Delaware	0.01	0.03	0.00	0.00	0.00	0.00
Florida	1.80	0.41	0.00	0.00	0.00	0.05
Georgia	1.16	0.24	0.01	0.00	0.00	0.03
Hawaii	0.18	0.02	0.00	0.00	0.00	**
Idaho	1.54	1.20	0.00	0.05	0.00	0.05
Illinois	1.20	0.35	0.13	0.01	0.00	0.04
Indiana	0.57	0.43	0.10	0.01	0.00	0.05
Iowa	2.93	0.64	0.47	0.05	0.00	0.05
Kansas	5.56	0.31	0.05	0.02	0.00	0.04
Kentucky	2.12	0.35	0.01	0.00	0.00	0.08
Louisiana	0.93	0.14	0.00	0.00	0.00	0.02
Maine	0.03	0.12	0.00	0.00	0.00	0.00
Maryland	0.09	0.24	0.00	0.00	0.00	0.02
Massachusetts	0.01	0.06	0.00	0.00	0.00	0.01
Michigan	0.35	0.86	0.03	0.01	0.00	0.05
Minnesota	1.33	1.57	0.18	0.03	0.00	0.05
Mississippi	1.11	0.10	0.01	0.00	0.00	0.03
Missouri	4.33	0.43	0.09	0.01	0.00	0.07
Montana	3.30	0.06	0.01	0.06	0.00	0.06
Nebraska	6.03	0.21	0.09	0.02	0.00	0.04
Nevada	0.62	0.09	0.00	0.02	0.00	0.01
New Hampshire	0.01	0.05	0.00	0.00	0.00	0.00
New Jersey	0.02	0.05	0.00	0.00	0.00	0.02
New Mexico	1.46	0.81	0.00	0.04	0.00	0.03
New York	0.23	1.98	0.00	0.01	0.00	0.04
North Carolina	0.85	0.20	0.30	0.00	0.00	0.03
North Dakota	2.37	0.13	0.00	0.02	0.00	0.03
Ohio	0.69	0.76	0.05	0.02	0.00	0.06
Oklahoma	5.02	0.22	0.07	0.01	0.00	0.08
Oregon	1.59	0.33	0.00	0.04	0.00	0.06
Pennsylvania	0.45	1.82	0.03	0.01	0.00	0.05
Rhode Island	0.00	0.01	0.00	0.00	0.00	0.00
South Carolina	0.43	0.06	0.01	0.00	0.00	0.02
South Dakota	4.47	0.28	0.04	0.07	0.00	0.04
Tennessee	2.02	0.28	0.01	0.00	0.01	0.07
Texas	12.88	0.89	0.03	0.19	0.13	0.20
Utah	0.97	0.32	0.02	0.07	0.00	0.04
Vermont	0.03	0.45	0.00	0.00	0.00	0.01
Virginia	1.39	0.35	0.01	0.01	0.00	0.04
Washington	0.82	0.82	0.00	0.01	0.00	0.05
West Virginia	0.41	0.05	0.00	0.01	0.00	0.01
Wisconsin	0.81	3.89	0.02	0.01	0.00	0.04
Wyoming	1.83	0.01	0.00	0.09	0.00	0.04
Total	82.65	26.93	1.86	1.17	0.21	2.00

Appendix Table A-5 Cattle population categories used for estimating methane emissions

Dairy cattle	Beef cattle
Calves	Calves
Heifer replacements	Heifer replacements
Cows	Heifer and steer stockers
	Animals in feedlots
	Cows
	Bulls

Source: EPA (2003a).

Appendix Table A-6 Dairy lactation by region

Year	California	West	Northern Great Plains	South Central	Northeast	Midwest	Southeast
				(lbs/year)/cow			
1990	18,443	17,293	13,431	13,399	14,557	14,214	12,852
1991	18,522	17,615	13,525	13,216	14,985	14,446	13,053
1992	18,709	18,083	13,998	13,656	15,688	14,999	13,451
1993	18,839	18,253	14,090	14,027	15,602	15,086	13,739
1994	20,190	18,802	14,686	14,395	15,732	15,276	14,111
1995	19,559	18,708	14,807	14,294	16,254	15,680	14,318
1996	19,148	19,076	15,040	14,402	16,271	15,651	14,232
1997	19,815	19,537	15,396	14,330	16,519	16,116	14,517
1998	19,461	19,814	15,922	14,722	16,865	16,676	14,404
1999	20,763	20,495	16,378	14,986	17,271	16,966	14,860
2000	21,134	20,782	17,297	15,314	17,484	17,426	15,196
2001	20,898	20,656	17,347	14,813	17,602	17,218	15,303

Source: USDA NASS Milk Production (2002, 2001, 2000); USDA NASS Cattle (1999, 1995); EPA (2003a).

Appendix Table A-7 Typical livestock weights

Cattle type	Typical weights
	lbs.
Beef replacement heifer data	
Replacement weight at 15 months	715
Replacement weight at 24 months	1,078
Mature weight at 36 months	1,172
Dairy replacement heifer data	
Replacement weight at 15 months	800
Replacement weight at 24 months	1,225
Mature weight at 36 months	1,350
Stockers data – grazing/forage based only	
Steer weight gain/month to 12 months	45
Steer weight gain/month to 24 months	35
Heifer weight gain/month to 12 months	35
Heifer weight gain/month to 24 months	30

Source: Feedstuffs (1998), Western Dairyman (1998), Johnson (1999), NRC (1999), EPA (2003a).

Appendix Table A-8 U.S. feedlot placement in 2001

Weight when placed	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
	Number of animals placed, 1,000 head												
< 600 lbs.	499	336	329	334	499	419	469	510	528	891	655	418	5,887
600 - 700 lbs.	716	402	414	384	509	442	444	472	448	758	588	475	6,052
700 - 800 lbs.	664	517	614	494	799	631	606	667	561	592	381	413	6,939
> 800 lbs.	384	325	485	339	565	473	467	555	604	461	284	272	5,214
Total	**2,263**	**1,580**	**1,842**	**1,551**	**2,372**	**1,965**	**1,986**	**2,204**	**2,141**	**2,702**	**1,908**	**1,578**	**22,184**

Source: USDA NASS Cattle on feed (2002, 2001, 2000); USDA NASS Cattle (1999, 1995); EPA (2003a).
Note: Totals may not sum due to independent rounding.

Appendix Table A-9 Regional estimates of digestible energy and methane conversion rates for enteric fermentation

Livestock Category	Data	California	West	Northern Great Plains	South Central	North-east	Midwest	Southeast
Beef replacement heifer	DE[a]	65	59	66	64	65	65	64
	Ym[b]	6.50%	6.50%	6.50%	6.50%	6.50%	6.50%	6.50%
	Pop.[c]	3%	10%	31%	23%	2%	14%	17%
Dairy replacement heifer	DE	66	66	66	64	68	66	66
	Ym	5.90%	5.90%	5.60%	6.40%	6.30%	5.60%	6.90%
	Pop.	18%	12%	5%	4%	18%	36%	7%
Steer stockers	DE	65	59	66	64	65	65	64
	Ym	6.50%	6.50%	6.50%	6.50%	6.50%	6.50%	6.50%
	Pop.	4%	8%	42%	22%	2%	18%	5%
Heifer stockers	DE	65	59	66	64	65	65	64
	Ym	6.50%	6.50%	6.50%	6.50%	6.50%	6.50%	6.50%
	Pop.	2%	7%	50%	22%	1%	15%	4%
Steer feedlot	DE	85	85	85	85	85	85	85
	Ym	3.00%	3.00%	3.00%	3.00%	3.00%	3.00%	3.00%
	Pop.	3%	8%	48%	24%	1%	16%	1%
Heifer feedlot	DE	85	85	85	85	85	85	85
	Ym	3.00%	3.00%	3.00%	3.00%	3.00%	3.00%	3.00%
	Pop.	3%	8%	48%	24%	1%	16%	1%
Beef cows	DE	63	57	64	62	63	63	62
	Ym	6.50%	6.50%	6.50%	6.50%	6.50%	6.50%	6.50%
	Pop.	2%	8%	28%	26%	2%	14%	19%
Dairy cows	DE	69	66	69	68	69	69	68
	Ym	4.80%	5.80%	5.80%	5.70%	5.80%	5.80%	5.60%
	Pop.	17%	13%	5%	6%	18%	33%	8%
Steer step-up	DE	73	73	73	73	73	73	73
	Ym	4.80%	4.80%	4.80%	4.80%	4.80%	4.80%	4.80%
Heifer step-up	DE	73	73	73	73	73	73	73
	Ym	4.80%	4.80%	4.80%	4.80%	4.80%	4.80%	4.80%

Source: EPA (2003a).

[a] Digestible Energy in units of percent GE (MJ/Day).

[b] Methane Conversion Rate is the fraction of GE in feed converted to methane.

[c] Percent of each subcategory population present in each region.

Appendix Table A-10 DE values and representative regional diets (percent of diet for each region) for the

Feed	Source of TDN (NRC 2000)	Unweighted TDN or DE	California	West	Northern	South Central	Northeast	Midwest	Southeast
Alfalfa hay	Table 11-1, feed #4	59.6%	65.0%	30.0%	30.0%	29.0%	12.0%	30.0%	
Barley	Table 11-1, feed #12	86.3%	10.0%	15.0%					
Bermuda	Table 11-1, feed #17	48.5%							35.0%
Bermuda hay	Table 11-1, feed #17	48.5%				40.0%			
Corn	Table 11-1, feed #38	88.1%	10.0%	10.0%	25.0%	11.0%	13.0%	13.0%	
Corn silage	Table 11-1, feed #39	71.2%			25.0%		20.0%	20.0%	
Cotton seed meal	Table 11-1, feed #42	74.4%				7.0%			
Grass hay	Table 1a, feed #129, 147, 148	53.7%		40.0%				30.0%	
Orchard	Table 11-1, feed #61	53.5%							40.0%
Soybean meal supplement	Table 11-1, feed #70	83.1%		5.0%	5.0%				5.0%
Sorghum	Table 11-1, feed #67	81.3%							20.0%
Soybean hulls	Table 11-1, feed #69	76.4%						7.0%	
Timothy hay	Table 11-1, feed #77	55.5%					50.0%		
Whole cotton seed	Table 11-1, feed #41	89.2%	5.0%				5.0%		
Wheat middlings	Table 1a, feed #433	83.0%			15.0%	13.0%			
Wheat	Table 11-1, feed #83	87.2%	10.0%						
Weighted total			**65.2%**	**65.1%**	**62.4%**	**65.0%**	**74.3%**	**58.8%**	**69.3%**

Sources: EPA (2003a); representative regional diets, Donovan (1999), NRC (2000).

Appendix Table A-11 Percent of each diet that is supplemental, and the resulting DE values for each region

Region	Percent supplement	Percent forage	Calculated weighted average DE
West	10	90	59.2%
Northeast	15	85	64.7%
South Central	10	90	64.4%
Midwest	15	85	64.7%
Northern Great Plains	15	85	66.1%
Southeast	5	95	64.4%
California	5	95	64.9%

Sources: EPA (2003a); percent of diet that is supplemental, Donovan (1999).

Appendix Table A-12 Methane emissions from cattle from enteric fermentation

Cattle type	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
						Gg						
Dairy	1,375	1,378	1,375	1,316	1,314	1,320	1,254	1,255	1,251	1,266	1,284	1,282
Cows	1,142	1,148	1,143	1,082	1,082	1,088	1,024	1,028	1,026	1,038	1,059	1,055
Replacements 7-11 months	49	49	49	49	49	49	48	48	48	48	48	48
Replacements 12-23 months	184	181	183	185	183	183	181	179	177	180	177	179
Beef	3,961	3,920	4,031	4,070	4,147	4,272	4,227	4,124	4,046	4,035	3,976	3,936
Cows	2,428	2,432	2,468	2,494	2,585	2,628	2,638	2,574	2,531	2,520	2,506	2,492
Replacements 7-11 months	52	54	57	60	62	61	60	56	54	53	53	54
Replacements 12-23 months	190	196	203	216	229	232	225	216	206	198	198	200
Steer stockers	430	402	464	482	435	479	455	430	418	393	369	372
Heifer stockers	231	220	233	240	231	249	239	241	236	227	213	215
Feedlot cattle	413	397	384	353	374	383	371	375	378	420	416	384
Bulls	218	220	222	224	231	239	239	233	223	224	220	219
Total	5,336	5,298	5,406	5,385	5,461	5,591	5,481	5,379	5,297	5,300	5,260	5,218

Source: EPA (2003a).
Note: Totals may not sum due to independent rounding.

Appendix Table A-13 IPCC emission factors for livestock

	Emission factors
	kg CH_4/head/year
Bulls	100
Calves	0
Swine	1.5
Sheep	8
Goats	5
Horses	18

Source: IPCC (2000).

Appendix Table A-14 Methane emissions from manure management in 2001 by State and animal

	Beef cattle	Dairy cattle	Poultry	Swine	Goats	Horses	Sheep	Total
				Tg CO_2 eq.				
Alabama	0.0541	0.0152	0.2549	0.0694	0.0001	0.0111	0.0000	0.4048
Alaska	0.0004	0.0014	0.0033	0.0000	0.0000	0.0000	0.0000	0.0052
Arizona	0.0300	0.6822	0.0033	0.0534	0.0007	0.0099	0.0006	0.7801
Arkansas	0.0701	0.0173	0.1072	0.2609	0.0001	0.0105	0.0000	0.4662
California	0.0974	5.5804	0.1259	0.0417	0.0002	0.0295	0.0043	5.8794
Colorado	0.1091	0.2423	0.0636	0.1637	0.0000	0.0213	0.0024	0.6024
Connecticut	0.0008	0.0179	0.0066	0.0004	0.0000	0.0018	0.0000	0.0275
Delaware	0.0005	0.0055	0.0189	0.0065	0.0000	0.0008	0.0000	0.0323
Florida	0.0677	0.4188	0.1713	0.0051	0.0008	0.0143	0.0000	0.6780
Georgia	0.0457	0.0808	0.4459	0.0314	0.0002	0.0092	0.0000	0.6132
Hawaii	0.0072	0.0303	0.0050	0.0078	0.0000	0.0000	0.0000	0.0504
Idaho	0.0653	1.2628	0.0179	0.0039	0.0000	0.0155	0.0013	1.3667
Illinois	0.0480	0.0902	0.0061	1.0032	0.0000	0.0135	0.0004	1.1616
Indiana	0.0242	0.0949	0.0304	0.7401	0.0000	0.0153	0.0002	0.9051
Iowa	0.1243	0.1372	0.0333	4.5277	0.0000	0.0158	0.0012	4.8395
Kansas	0.2326	0.0757	0.0046	0.4351	0.0000	0.0138	0.0008	0.7627
Kentucky	0.0834	0.0362	0.0273	0.0287	0.0001	0.0250	0.0000	0.2008
Louisiana	0.0334	0.0385	0.0530	0.0039	0.0001	0.0079	0.0000	0.1368
Maine	0.0013	0.0159	0.0158	0.0001	0.0000	0.0015	0.0000	0.0345
Maryland	0.0041	0.0478	0.0275	0.0125	0.0000	0.0059	0.0000	0.0978
Massachusetts	0.0006	0.0104	0.0006	0.0027	0.0000	0.0025	0.0000	0.0168
Michigan	0.0181	0.2959	0.0175	0.1962	0.0001	0.0173	0.0003	0.5453
Minnesota	0.0581	0.3029	0.0374	1.2267	0.0000	0.0146	0.0007	1.6404
Mississippi	0.0422	0.0220	0.2249	0.1267	0.0001	0.0082	0.0000	0.4242
Missouri	0.1596	0.1041	0.0188	0.7291	0.0001	0.0224	0.0004	1.0344
Montana	0.1054	0.0307	0.0067	0.0325	0.0000	0.0186	0.0014	0.1954
Nebraska	0.2420	0.0502	0.0171	0.6714	0.0000	0.0120	0.0007	0.9933
Nevada	0.0249	0.1017	0.0033	0.0001	0.0000	0.0037	0.0007	0.1344
New Hampshire	0.0005	0.0085	0.0004	0.0004	0.0000	0.0012	0.0000	0.0109
New Jersey	0.0009	0.0085	0.0043	0.0022	0.0000	0.0059	0.0000	0.0218
New Mexico	0.0605	1.0896	0.0033	0.0000	0.0005	0.0101	0.0018	1.1658
New York	0.0125	0.3919	0.0085	0.0124	0.0001	0.0125	0.0003	0.4381
North Carolina	0.0336	0.0299	0.2805	3.9581	0.0001	0.0107	0.0000	4.3129
North Dakota	0.0758	0.0196	0.0042	0.0285	0.0000	0.0092	0.0004	0.1377
Ohio	0.0310	0.1698	0.0261	0.3025	0.0001	0.0199	0.0008	0.5503
Oklahoma	0.1858	0.1930	0.1078	1.0173	0.0003	0.0244	0.0004	1.5291
Oregon	0.0628	0.1876	0.0313	0.0024	0.0001	0.0178	0.0013	0.3034
Pennsylvania	0.0233	0.2387	0.0303	0.2372	0.0001	0.0170	0.0004	0.5470
Rhode Island	0.0001	0.0004	0.0000	0.0004	0.0000	0.0003	0.0000	0.0012
South Carolina	0.0167	0.0150	0.1369	0.1314	0.0001	0.0060	0.0000	0.3062
South Dakota	0.1500	0.0599	0.0057	0.2815	0.0000	0.0135	0.0017	0.5124
Tennessee	0.0802	0.0350	0.0174	0.0163	0.0003	0.0232	0.0000	0.1724
Texas	0.5203	1.0565	0.1408	0.3653	0.0099	0.0631	0.0092	2.1651
Utah	0.0385	0.2868	0.0622	0.1393	0.0000	0.0128	0.0020	0.5417
Vermont	0.0021	0.0826	0.0004	0.0000	0.0000	0.0023	0.0000	0.0875
Virginia	0.0568	0.0419	0.0405	0.1490	0.0001	0.0131	0.0005	0.3018
Washington	0.0377	0.6253	0.0271	0.0039	0.0000	0.0153	0.0003	0.7096
West Virginia	0.0152	0.0074	0.0118	0.0019	0.0000	0.0044	0.0001	0.0409
Wisconsin	0.0410	0.8358	0.0083	0.1142	0.0001	0.0137	0.0004	1.0134
Wyoming	0.0613	0.0073	0.0002	0.0252	0.0000	0.0132	0.0025	0.1097

Appendix Table A-15 Nitrous oxide emissions from manure management in 2001 by State and animal

	Beef cattle	Dairy cattle	Poultry	Swine	Goats	Horses	Sheep	Total
				Tg CO_2 eq.				
Alabama	0.0017	0.0029	0.6500	0.0013	0.0002	0.0036	0.0002	0.6598
Alaska	0.0000	0.0004	0.0005	0.0000	0.0000	0.0000	0.0002	0.0011
Arizona	0.1292	0.0314	0.0005	0.0010	0.0003	0.0032	0.0000	0.1656
Arkansas	0.0047	0.0061	0.9226	0.0047	0.0001	0.0034	0.0002	0.9419
California	0.1932	0.5174	0.1976	0.0008	0.0003	0.0095	0.0002	0.9190
Colorado	0.5279	0.0358	0.0252	0.0068	0.0001	0.0069	0.0070	0.6097
Connecticut	0.0001	0.0087	0.0039	0.0000	0.0000	0.0006	0.0000	0.0133
Delaware	0.0001	0.0030	0.1647	0.0002	0.0000	0.0003	0.0002	0.1685
Florida	0.0010	0.0280	0.0831	0.0001	0.0002	0.0046	0.0002	0.1171
Georgia	0.0013	0.0131	0.8126	0.0021	0.0003	0.0030	0.0002	0.8326
Hawaii	0.0002	0.0024	0.0012	0.0001	0.0000	0.0000	0.0002	0.0041
Idaho	0.1395	0.1417	0.0007	0.0001	0.0001	0.0050	0.0000	0.2871
Illinois	0.0923	0.0823	0.0214	0.0308	0.0001	0.0043	0.0004	0.2316
Indiana	0.0472	0.0884	0.1740	0.0236	0.0001	0.0049	0.0000	0.3382
Iowa	0.4507	0.1398	0.1484	0.1059	0.0001	0.0051	0.0024	0.8524
Kansas	1.0517	0.0745	0.0251	0.0122	0.0001	0.0044	0.0010	1.1690
Kentucky	0.0086	0.0242	0.1664	0.0028	0.0001	0.0081	0.0002	0.2104
Louisiana	0.0008	0.0081	0.0623	0.0001	0.0001	0.0025	0.0002	0.0741
Maine	0.0001	0.0139	0.0667	0.0000	0.0000	0.0005	0.0000	0.0812
Maryland	0.0043	0.0282	0.1896	0.0004	0.0000	0.0019	0.0002	0.2246
Massachusetts	0.0001	0.0074	0.0008	0.0001	0.0000	0.0008	0.0000	0.0091
Michigan	0.0816	0.1790	0.0960	0.0071	0.0001	0.0056	0.0000	0.3693
Minnesota	0.1245	0.3269	0.2991	0.0436	0.0001	0.0047	0.0017	0.8005
Mississippi	0.0015	0.0052	0.4907	0.0021	0.0002	0.0027	0.0002	0.5026
Missouri	0.0279	0.0953	0.1482	0.0213	0.0002	0.0072	0.0000	0.3001
Montana	0.0258	0.0108	0.0003	0.0013	0.0000	0.0060	0.0006	0.0447
Nebraska	1.0946	0.0462	0.0403	0.0209	0.0000	0.0039	0.0012	1.2071
Nevada	0.0107	0.0093	0.0005	0.0000	0.0000	0.0012	0.0000	0.0218
New Hampshire	0.0000	0.0064	0.0002	0.0000	0.0000	0.0004	0.0000	0.0071
New Jersey	0.0009	0.0055	0.0029	0.0001	0.0000	0.0019	0.0002	0.0114
New Mexico	0.0494	0.0581	0.0005	0.0000	0.0004	0.0033	0.0000	0.1117
New York	0.0129	0.2204	0.0093	0.0005	0.0001	0.0040	0.0000	0.2471
North Carolina	0.0021	0.0105	0.7124	0.0661	0.0003	0.0035	0.0002	0.7952
North Dakota	0.0258	0.0260	0.0098	0.0011	0.0001	0.0030	0.0011	0.0667
Ohio	0.0816	0.1607	0.0955	0.0096	0.0002	0.0064	0.0009	0.3548
Oklahoma	0.1781	0.0406	0.1695	0.0170	0.0003	0.0079	0.0000	0.4134
Oregon	0.0236	0.0334	0.0869	0.0001	0.0001	0.0057	0.0007	0.1505
Pennsylvania	0.0322	0.2196	0.1706	0.0079	0.0002	0.0055	0.0000	0.4359
Rhode Island	0.0000	0.0006	0.0000	0.0000	0.0000	0.0001	0.0000	0.0007
South Carolina	0.0017	0.0028	0.1826	0.0023	0.0002	0.0019	0.0002	0.1918
South Dakota	0.1567	0.0572	0.0313	0.0095	0.0000	0.0044	0.0021	0.2611
Tennessee	0.0043	0.0170	0.1279	0.0015	0.0004	0.0075	0.0002	0.1587
Texas	1.2619	0.1099	0.4067	0.0059	0.0105	0.0204	0.0031	1.8182
Utah	0.0150	0.0446	0.0250	0.0048	0.0000	0.0041	0.0000	0.0936
Vermont	0.0001	0.0510	0.0005	0.0000	0.0000	0.0007	0.0000	0.0524
Virginia	0.0129	0.0209	0.3153	0.0029	0.0002	0.0042	0.0002	0.3565
Washington	0.1095	0.0734	0.0674	0.0001	0.0001	0.0049	0.0000	0.2554
West Virginia	0.0030	0.0055	0.0825	0.0001	0.0001	0.0014	0.0000	0.0925
Wisconsin	0.0687	0.8231	0.0265	0.0039	0.0002	0.0044	0.0000	0.9268
Wyoming	0.0343	0.0016	0.0000	0.0010	0.0001	0.0043	0.0021	0.0434

Appendix Table A-16 Waste characteristics data

Livestock	Average TAM	Source	Nitrogen, N_{ex}	Source	Max Methane Generation Potential, B_o	Source	Volatile Solids VS	Source
	kg		*kg/day per 1,000 kg mass*		*m^3 CH_4/kg VS added*		*kg/day per 1,000 kg mass*	
Dairy cow	604	Safley 2000	0.44	USDA NRCS 1996	0.24	Morris 1976	Table M-3	Peterson et al., 2002
Dairy heifer	476	Safley 2000	0.31	USDA NRCS 1996	0.17	Bryant et al. 1976	Table M-3	Peterson et al., 2002
Feedlot steers	420	USDA NRCS 1996	0.3	USDA NRCS 1996	0.33	Hashimoto 1981	Table M-3	Peterson et al., 2002
Feedlot heifers	420	USDA NRCS 1996	0.3	USDA NRCS 1996	0.33	Hashimoto 1981	Table M-3	Peterson et al., 2002
NOF bulls	750	Safley 2000	0.31	USDA NRCS 1996	0.17	Hashimoto 1981	6.04	USDA NRCS 1996
NOF calves	159	USDA NRCS 1998	0.3	USDA NRCS 1996	0.17	Hashimoto 1981	6.41	USDA NRCS 1996
NOF heifers	420	USDA NRCS 1996	0.31	USDA NRCS 1996	0.17	Hashimoto 1981	Table M-3	Peterson et al., 2002
NOF steers	318	Safley 2000	0.31	USDA NRCS 1996	0.17	Hashimoto 1981	Table M-3	Peterson et al., 2002
NOF cows	590	Safley 2000	0.33	USDA NRCS 1996	0.17	Hashimoto 1981	Table M-3	Peterson et al., 2002
Market swine <60 lbs.	15.88	Safley 2000	0.6	USDA NRCS 1996	0.48	Hashimoto 1984	8.8	USDA NRCS 1996
Market swine 60-119 lbs.	40.6	Safley 2000	0.42	USDA NRCS 1996	0.48	Hashimoto 1985	5.4	USDA NRCS 1996
Market swine 120-179 lbs.	67.82	Safley 2000	0.42	USDA NRCS 1996	0.48	Hashimoto 1984	5.4	USDA NRCS 1996
Market swine >180 lbs.	90.75	Safley 2000	0.42	USDA NRCS 1996	0.48	Hashimoto 1984	5.4	USDA NRCS 1996
Breeding swine	198	Safley 2000	0.24	USDA NRCS 1996	0.48	Hashimoto 1984	2.6	USDA NRCS 1996
Sheep	27	ASAE 1999	0.42	ASAE1999	NA	NA	NA	NA
Goats	64	ASAE 1999	0.45	ASAE 1999	NA	NA	NA	NA
Horses	450	ASAE 1999	0.3	ASAE 1999	NA	NA	NA	NA
Hens >/= 1 yr	1.8	ASAE 1999	0.83	USDA NRCS 1996	0.39	Hill 1982	10.8	USDA NRCS 1996
Pullets	1.8	ASAE 1999	0.62	USDA NRCS 1996	0.39	Hill 1982	9.7	USDA NRCS 1996
Other chickens	1.8	ASAE 1999	0.83	USDA NRCS 1996	0.39	Hill 1982	10.8	USDA NRCS 1996
Broilers	0.9	ASAE 1999	1.1	USDA NRCS 1996	0.36	Hill 1984	15	USDA NRCS 1996
Turkeys	6.8	ASAE 1999	0.74	USDA NRCS 1996	0.36	Hill 1984	9.7	USDA NRCS 1996

Source: EPA (2003a).

Appendix Table A-17 State volatile solids production rates for 2001

	Dairy cow	Dairy heifer	Cow (not on feed)	Heifer (not on feed)	Steer (not on feed)	Feedlot heifer	Feedlot steer
				kg/day/1000 kg mass			
Alabama	8.56	6.82	6.74	7.16	7.47	3.33	3.26
Alaska	10.71	6.82	8.71	9.42	9.87	3.33	3.26
Arizona	10.71	6.82	8.71	9.43	9.87	3.33	3.26
Arkansas	8.06	7.57	6.72	7.13	7.45	3.36	3.3
California	9.36	6.82	6.57	6.95	7.27	3.32	3.26
Colorado	8.33	6.82	6.19	6.51	6.82	3.35	3.28
Connecticut	8.41	6.14	6.62	7.03	7.33	3.4	3.33
Delaware	8.41	6.14	6.62	7.01	7.33	3.4	3.33
Florida	8.56	6.82	6.74	7.17	7.47	3.33	3.26
Georgia	8.56	6.82	6.74	7.16	7.47	3.33	3.26
Hawaii	10.71	6.82	8.71	9.42	9.87	3.33	3.26
Idaho	10.71	6.82	8.71	9.4	9.87	3.33	3.26
Illinois	8.29	6.82	6.63	7.01	7.34	3.39	3.32
Indiana	8.29	6.82	6.63	7.01	7.34	3.39	3.32
Iowa	8.29	6.82	6.63	7	7.34	3.39	3.32
Kansas	8.33	6.82	6.19	6.51	6.82	3.35	3.28
Kentucky	8.56	6.82	6.74	7.16	7.47	3.33	3.26
Louisiana	8.06	7.57	6.72	7.14	7.45	3.36	3.3
Maine	8.41	6.14	6.62	7.04	7.33	3.4	3.33
Maryland	8.41	6.14	6.62	7.02	7.33	3.4	3.33
Massachusetts	8.41	6.14	6.62	7.02	7.33	3.4	3.33
Michigan	8.29	6.82	6.63	7.02	7.34	3.39	3.32
Minnesota	8.29	6.82	6.63	7.01	7.34	3.39	3.32
Mississippi	8.56	6.82	6.74	7.17	7.47	3.33	3.26
Missouri	8.29	6.82	6.63	7.02	7.34	3.39	3.32
Montana	8.33	6.82	6.19	6.54	6.82	3.35	3.28
Nebraska	8.33	6.82	6.19	6.51	6.82	3.35	3.28
Nevada	10.71	6.82	8.71	9.41	9.87	3.33	3.26
New Hampshire	8.41	6.14	6.62	7.04	7.33	3.4	3.33
New Jersey	8.41	6.14	6.62	7.03	7.33	3.4	3.33
New Mexico	10.71	6.82	8.71	9.4	9.87	3.33	3.26
New York	8.41	6.14	6.62	7.01	7.33	3.4	3.33
North Carolina	8.56	6.82	6.74	7.17	7.47	3.33	3.26
North Dakota	8.33	6.82	6.19	6.52	6.82	3.35	3.28
Ohio	8.29	6.82	6.63	7.02	7.34	3.39	3.32
Oklahoma	8.06	7.57	6.72	7.11	7.45	3.36	3.3
Oregon	10.71	6.82	8.71	9.42	9.87	3.33	3.26
Pennsylvania	8.41	6.14	6.62	7.02	7.33	3.4	3.33
Rhode Island	8.41	6.14	6.62	7.04	7.33	3.4	3.33
South Carolina	8.56	6.82	6.74	7.17	7.47	3.33	3.26
South Dakota	8.33	6.82	6.19	6.52	6.82	3.35	3.28
Tennessee	8.56	6.82	6.74	7.16	7.47	3.33	3.26
Texas	8.06	7.57	6.72	7.11	7.45	3.36	3.3
Utah	10.71	6.82	8.71	9.41	9.87	3.33	3.26
Vermont	8.41	6.14	6.62	7.02	7.33	3.4	3.33
Virginia	8.56	6.82	6.74	7.16	7.47	3.33	3.26
Washington	10.71	6.82	8.71	9.4	9.87	3.33	3.26
West Virginia	8.41	6.14	6.62	7.03	7.33	3.4	3.33
Wisconsin	8.29	6.82	6.63	7.01	7.34	3.39	3.32
Wyoming	8.33	6.82	6.19	6.53	6.82	3.35	3.28

Source: Peterson et al. (2002); EPA (2003a).

Appendix Table A-18 State base methane emissions factors by waste management system for 2001

	Liquid/ slurry	Anaerobic lagoon	Deep pit
Alabama	0.3511	0.7663	0.3511
Alaska	0.1507	0.4845	0.1507
Arizona	0.4673	0.7918	0.4673
Arkansas	0.3760	0.7617	0.3760
California	0.3630	0.7554	0.3630
Colorado	0.2297	0.6668	0.2297
Connecticut	0.2545	0.6763	0.2545
Delaware	0.2823	0.6862	0.2823
Florida	0.5195	0.7935	0.5195
Georgia	0.3263	0.6578	0.3263
Hawaii	0.5973	0.7728	0.5973
Idaho	0.2311	0.6741	0.2311
Illinois	0.2935	0.7202	0.2935
Indiana	0.2792	0.7097	0.2792
Iowa	0.2634	0.6986	0.2634
Kansas	0.3401	0.7493	0.3401
Kentucky	0.2726	0.6301	0.2726
Louisiana	0.4542	0.7860	0.4542
Maine	0.2119	0.6390	0.2119
Maryland	0.2847	0.7190	0.2847
Massachusetts	0.2448	0.6871	0.2448
Michigan	0.2395	0.6751	0.2395
Minnesota	0.2407	0.6785	0.2407
Mississippi	0.4015	0.7722	0.4015
Missouri	0.3252	0.7398	0.3252
Montana	0.2153	0.6508	0.2153
Nebraska	0.2815	0.7166	0.2815
Nevada	0.2597	0.7009	0.2597
New Hampshire	0.2191	0.6501	0.2191
New Jersey	0.2778	0.7160	0.2778
New Mexico	0.3210	0.7387	0.3210
New York	0.2307	0.6683	0.2307
North Carolina	0.3320	0.7473	0.3320
North Dakota	0.2256	0.6612	0.2256
Ohio	0.2652	0.6985	0.2652
Oklahoma	0.3962	0.7681	0.3962
Oregon	0.2120	0.6429	0.2120
Pennsylvania	0.2610	0.7000	0.2610
Rhode Island	0.2242	0.6032	0.2242
South Carolina	0.3804	0.7690	0.3804
South Dakota	0.2552	0.6970	0.2552
Tennessee	0.2796	0.6343	0.2796
Texas	0.4466	0.7817	0.4466
Utah	0.2681	0.7116	0.2681
Vermont	0.2134	0.6407	0.2134
Virginia	0.2816	0.7142	0.2816
Washington	0.2153	0.6498	0.2153
West Virginia	0.2613	0.6968	0.2613
Wisconsin	0.2353	0.6714	0.2353
Wyoming	0.2244	0.6635	0.2244

Source: EPA (2003a).

Appendix Table A-19 Additional nitrous oxide and methane emission factors

	CH_4	N_2O
Pasture	0.015	
Daily spread	0.005	
Solid Storage	0.015	
Dry lot	0.015	
Poultry with bedding	0.015	
Poultry without bedding	0.015	
Liquid systems		0.001
Dry Systems		0.02

Source: IPCC (2000).

Appendix Table A-20 State-weighted methane conversion factors for livestock waste emissions for 2001

	Beef feed-lot-heifer	Beef feed-lot-steer	Dairy cow	Dairy heifer	Swine-market	Swine-breeding	Layer	Broiler	Turkey
	Proportion of waste attributed to livestock category								
Alabama	0.0200	0.0170	0.1019	0.0189	0.4889	0.4912	0.3290	0.0150	0.0150
Alaska	0.0170	0.0170	0.1652	0.0165	0.0150	0.0150	0.1324	0.0150	0.0150
Arizona	0.0169	0.0166	0.6180	0.0165	0.5272	0.5272	0.4842	0.0150	0.0150
Arkansas	0.0199	0.0199	0.0754	0.0188	0.5499	0.5539	0.0150	0.0150	0.0150
California	0.0195	0.0197	0.5167	0.0184	0.5056	0.5022	0.1061	0.0150	0.0150
Colorado	0.0159	0.0159	0.4365	0.0157	0.2839	0.2835	0.4032	0.0150	0.0150
Connecticut	0.0175	0.0175	0.1082	0.0170	0.1443	0.1311	0.0492	0.0150	0.0150
Delaware	0.0181	0.0181	0.0967	0.0175	0.3315	0.3315	0.0329	0.0150	0.0150
Florida	0.0220	0.0220	0.4274	0.0204	0.2150	0.2160	0.3413	0.0150	0.0150
Georgia	0.0200	0.0200	0.1465	0.0188	0.0019	0.4957	0.3258	0.0150	0.0150
Hawaii	0.0228	0.0228	0.5474	0.0210	0.3966	0.3966	0.2045	0.0150	0.0150
Idaho	0.0159	0.0159	0.4509	0.0157	0.2101	0.2093	0.4043	0.0150	0.0150
Illinois	0.0167	0.0167	0.1219	0.0164	0.3354	0.3355	0.0293	0.0150	0.0150
Indiana	0.0167	0.0167	0.1002	0.0164	0.3230	0.3233	0.0150	0.0150	0.0150
Iowa	0.0166	0.0166	0.1009	0.0163	0.4187	0.4195	0.0150	0.0150	0.0150
Kansas	0.0170	0.0170	0.1236	0.0167	0.3638	0.3637	0.0297	0.0150	0.0150
Kentucky	0.0182	0.0182	0.0421	0.0175	0.0016	0.4517	0.0510	0.0150	0.0150
Louisiana	0.0209	0.0209	0.1121	0.0196	0.2039	0.2035	0.4777	0.0150	0.0150
Maine	0.0171	0.0171	0.0642	0.0167	0.0150	0.0150	0.0464	0.0150	0.0150
Maryland	0.0178	0.0177	0.0916	0.0172	0.2957	0.2956	0.0507	0.0150	0.0150
Massachusetts	0.0174	0.0175	0.0774	0.0169	0.1974	0.1968	0.0486	0.0150	0.0150
Michigan	0.0164	0.0164	0.1589	0.0162	0.2937	0.2927	0.0284	0.0150	0.0150
Minnesota	0.0164	0.0164	0.0916	0.0162	0.3064	0.3062	0.0150	0.0150	0.0150
Mississippi	0.0202	0.0202	0.0939	0.0190	0.5618	0.5622	0.4697	0.0150	0.0150
Missouri	0.0169	0.0169	0.1109	0.0166	0.3551	0.3550	0.0150	0.0150	0.0150
Montana	0.0159	0.0159	0.2602	0.0157	0.2625	0.2625	0.3985	0.0150	0.0150
Nebraska	0.0167	0.0167	0.1065	0.0164	0.3279	0.3275	0.0291	0.0150	0.0150
Nevada	0.0160	0.0160	0.5146	0.0157	0.0150	0.0150	0.0150	0.0150	0.0150
New Hampshire	0.0172	0.0172	0.0733	0.0167	0.1228	0.1221	0.0470	0.0150	0.0150
New Jersey	0.0178	0.0178	0.0831	0.0172	0.1894	0.1911	0.0500	0.0150	0.0150
New Mexico	0.0162	0.0162	0.5294	0.0159	0.0150	0.0150	0.4606	0.0150	0.0150
New York	0.0173	0.0173	0.0918	0.0168	0.2126	0.2119	0.0476	0.0150	0.0150
North Carolina	0.0182	0.0182	0.0656	0.0175	0.5841	0.5824	0.3209	0.0150	0.0150
North Dakota	0.0164	0.0164	0.0675	0.0161	0.2615	0.2618	0.0277	0.0150	0.0150
Ohio	0.0166	0.0166	0.1021	0.0163	0.3032	0.3034	0.0150	0.0150	0.0150
Oklahoma	0.0166	0.0166	0.3603	0.0162	0.5767	0.5811	0.4671	0.0150	0.0150
Oregon	0.0178	0.0178	0.2611	0.0172	0.1098	0.1096	0.1676	0.0150	0.0150
Pennsylvania	0.0175	0.0176	0.0603	0.0170	0.3196	0.3187	0.0150	0.0150	0.0150
Rhode Island	0.0176	0.0176	0.0394	0.0171	0.1953	0.1953	0.0143	0.0150	0.0150
South Carolina	0.0198	0.0198	0.1051	0.0187	0.5174	0.5149	0.4696	0.0150	0.0150
South Dakota	0.0165	0.0165	0.0949	0.0163	0.3058	0.3060	0.0285	0.0150	0.0150
Tennessee	0.0182	0.0182	0.0551	0.0176	0.0025	0.4238	0.0513	0.0150	0.0150
Texas	0.0167	0.0167	0.5148	0.0163	0.5367	0.5362	0.1075	0.0150	0.0150
Utah	0.0160	0.0160	0.3811	0.0158	0.3313	0.3290	0.4377	0.0150	0.0150
Vermont	0.0172	0.0172	0.0837	0.0167	0.0150	0.0150	0.0458	0.0150	0.0150
Virginia	0.0178	0.0177	0.0525	0.0172	0.4896	0.4901	0.0497	0.0150	0.0150
Washington	0.0178	0.0180	0.3206	0.0172	0.2133	0.2097	0.0888	0.0150	0.0150
West Virginia	0.0176	0.0176	0.0689	0.0171	0.2068	0.2062	0.0493	0.0150	0.0150
Wisconsin	0.0164	0.0164	0.1002	0.0162	0.2757	0.2753	0.0281	0.0150	0.0150
Wyoming	0.0159	0.0159	0.2385	0.0157	0.2863	0.2848	0.3997	0.0150	0.0150

Source: EPA (2003a).

Appendix Table A-21 Nitrogen in livestock waste deposited in pasture, range, and paddock

	Pasture, range, and paddock
	Gg
1990	4,148
1991	4,167
1992	4,256
1993	4,304
1994	4,412
1995	4,474
1996	4,467
1997	4,331
1998	4,244
1999	4,199
2000	4,136
2001	4,100

Source: EPA (2003a).

Appendix Table A-22 Definition of regions in the enteric fermentation model

Region	State
California	California
Midwest	Illinois
	Indiana
	Iowa
	Michigan
	Minnesota
	Missouri
	Ohio
	Wisconsin
Northeast	Connecticut
	Delaware
	Maine
	Maryland
	Massachusetts
	New Hampshire
	New Jersey
	New York
	Pennsylvania
	Rhode Island
	Vermont
	West Virginia
Northern Great Plains	Colorado
	Kansas
	Montana
	Nebraska
	North Dakota
	South Dakota
	Wyoming
South Central	Arkansas
	Louisiana
	Oklahoma
	Texas
Southeast	Alabama
	Florida
	Georgia
	Kentucky
	Mississippi
	North Carolina
	South Carolina
	Tennessee
	Virginia
West	Alaska
	Arizona
	Hawaii
	Idaho
	Nevada
	New Mexico
	Oregon
	Utah
	Washington

Source: EPA (2003b).

Appendix B: Cropland Agriculture

Appendix Table B-1 Coefficients for estimating methane and nitrous oxide emissions from residue burning

	Corn	Peanuts	Soy-beans	Barley	Wheat	Rice	Sugar-cane
Res/crop ratio	1	1	2.1	1.2	1.3	1.4	0.8
Fraction residue burned*	0.03	0.03	0.03	0.03	0.03	see below	0.03
Dry matter fraction	0.91	0.86	0.87	0.93	0.93	0.91	0.62
Burning efficiency	0.93	0.93	0.93	0.93	0.93	0.93	0.93
Combustion efficiency	0.88	0.88	0.88	0.88	0.88	0.88	0.88
Fraction carbon	0.4478	0.45	0.45	0.4485	0.4428	0.3806	0.4235
Fraction nitrogen	0.0058	0.0106	0.023	0.0077	0.0062	0.0072	0.004

	All Crops
CH_4 emissions factor	0.005
CH_4 global warming potential	21
N_2O emissions factor	0.007
N_2O global warming potential	310

*Percent of Rice Acreage Burned	2001
Arkansas	0.10
California	0.23
Florida	0.00
Louisiana	0.04
Mississippi	0.40
Missouri	0.05
Texas	0.00

Source: EPA (2003a and 2003b).

Appendix Table B-2 Total U.S. production of crops managed with burning, 1990-2001

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
	Million metric tons											
Wheat	74.29	53.89	67.14	65.22	63.17	59.40	61.98	67.53	69.33	62.57	60.76	53.28
Rice[1]	7.10	7.27	8.20	7.13	9.02	7.94	7.83	8.34	8.57	9.38	8.70	9.69
Sugar-cane	25.52	27.44	27.54	28.19	28.06	27.92	26.73	28.77	30.90	32.02	32.76	31.57
Corn	201.53	189.87	240.72	160.99	255.29	187.97	234.52	233.86	247.88	239.55	251.85	241.48
Barley	9.19	10.11	9.91	8.67	8.16	7.82	8.54	7.84	7.67	6.10	6.94	5.43
Soybeans	52.42	54.06	59.61	50.89	68.44	59.17	64.78	73.18	74.60	72.22	75.06	78.67
Peanuts	1.63	2.23	1.94	1.54	1.93	1.57	1.66	1.61	1.80	1.74	1.48	1.92
Total	371.70	344.88	415.06	322.61	434.07	351.80	406.04	421.12	440.74	423.58	437.55	422.05

Source: USDA NASS Field Crops (1994, 1998); USDA NASS Crop Production (2002).

[1] Rice production data for Florida were estimated from information provided by Smith (1999), Schueneman (1999b, 2001), and Deren (2002).

Appendix Table B-3 State production of crops managed with burning, 2001

	Corn	Peanuts	Soybeans	Barley	Wheat	Rice	Sugarcane
	1,000 bushels	*1,000 lbs.*	*1,000 bushels*	*1,000 bushels*	*1,000 bushels*	*1,000 cwt*	*1,000 tons*
Alaska	ND[1]	ND	ND	ND	ND	ND	ND
Alabama	16,050	547,250	4,725	0	3,360	0	0
Arkansas	26,825	0	91,200	0	50,440	101,312	0
Arizona	5,824	0	0	4,400	8,517	0	0
California	27,200	0	0	5,830	35,105	38,490	0
Colorado	149,800	0	0	8,560	69,168	0	0
Connecticut	ND	0	0	0	0	0	0
Delaware	23,652	0	7,839	2,002	3,477	0	0
Florida	2,262	261,450	261	0	369	501	16,472
Georgia	29,480	1,689,600	4,185	0	10,600	0	0
Hawaii	0	0	0	0	0	0	1,972
Iowa	1,664,400	0	480,480	0	972	0	0
Idaho	6,750	0	0	50,250	85,150	0	0
Illinois	1,649,200	0	477,900	0	43,920	0	0
Indiana	884,520	0	273,910	0	25,080	0	0
Kansas	387,350	0	87,360	400	328,000	0	0
Kentucky	156,200	0	48,800	680	23,760	0	0
Louisiana	45,436	0	20,130	0	8,000	30,014	14,850
Massachusetts	ND	0	0	0	0	0	0
Maryland	55,760	0	20,085	3,825	11,025	0	0
Maine	ND	0	0	1,820	0	0	0
Michigan	199,500	0	63,900	1,008	35,840	0	0
Minnesota	806,000	0	266,400	7,975	79,655	0	0
Missouri	345,800	0	186,200	0	41,040	12,317	0
Mississippi	50,050	0	36,960	0	11,700	16,445	0
Montana	1,924	0	0	29,520	96,570	0	0
North Carolina	78,125	369,000	43,200	1,206	18,330	0	0
North Dakota	81,075	0	71,740	79,750	292,400	0	0
Nebraska	1,139,250	0	222,950	180	59,200	0	0
New Hampshire	ND	0	0	0	0	0	0
New Jersey	7,392	0	3,131	216	1,215	0	0
New Mexico	8,280	66,700	0	0	8,160	0	0
Nevada	ND	0	0	90	270	0	0
New York	56,700	0	5,214	612	6,360	0	0
Ohio	437,460	0	187,780	380	60,300	0	0
Oklahoma	26,250	187,500	5,035	ND	122,100	0	0
Oregon	2,520	0	0	4,500	33,250	0	0
Pennsylvania	97,020	0	14,175	4,200	8,320	0	0
Rhode Island	ND	0	0	0	0	0	0
South Carolina	25,920	30,450	9,460	ND	9,030	0	0
South Dakota	370,600	0	138,570	4,056	76,766	0	0
Tennessee	81,840	0	35,700	0	18,360	0	0
Texas	167,560	855,000	5,670	ND	108,800	14,467	1,507
Utah	2,130	0	0	4,420	6,034	0	0
Virginia	40,590	232,500	17,280	3,750	10,200	0	0
Vermont	ND	0	0	0	0	0	0
Washington	10,450	0	0	21,000	132,580	0	0
Wisconsin	330,200	0	59,660	1,820	10,708	0	0
West Virginia	3,120	0	672	0	464	0	0
Wyoming	6,375	0	0	7,140	3,048	0	0
United States	9,506,840	4,239,450	2,890,572	249,590	1,957,643	213,045	34,801

Source: USDA NASS Crop Production (2002).

[1] ND = no data.

Appendix Table B-4 State estimates of methane emissions from agriculture burning by crop type , 2001

	Corn	Peanuts	Soybeans	Barley	Wheat	Rice	Sugarcane	Total
				Tg CO_2 eq.				
Alaska	ND[1]	ND	ND	ND	ND	ND	ND	ND
Alabama	0.0006	0.0003	0.0004	-[2]	0.0002	-	-	0.0014
Arkansas	0.0010	-	0.0070	-	0.0025	0.0255	-	0.0359
Arizona	0.0002	-	-	0.0002	0.0004	-	-	0.0008
California	0.0010	-	-	0.0002	0.0018	0.0220	-	0.0249
Colorado	0.0053	-	-	0.0003	0.0035	-	-	0.0091
Connecticut	ND	-	-	-	-	-	-	-
Delaware	0.0008	-	0.0006	0.0001	0.0002	-	-	0.0017
Florida	0.0001	0.0002	0.0000	-	0.0000	-	0.0108	0.0110
Georgia	0.0010	0.0010	0.0003	-	0.0005	-	-	0.0029
Hawaii	-	-	-	-	-	-	0.0013	0.0013
Iowa	0.0591	-	0.0369	-	0.0000	-	-	0.0960
Idaho	0.0002	-	-	0.0019	0.0043	-	-	0.0064
Illinois	0.0585	-	0.0367	-	0.0022	-	-	0.0974
Indiana	0.0314	-	0.0210	-	0.0013	-	-	0.0537
Kansas	0.0137	-	0.0067	0.0000	0.0164	-	-	0.0368
Kentucky	0.0055	-	0.0037	0.0000	0.0012	-	-	0.0105
Louisiana	0.0016	-	0.0015	-	0.0004	0.0030	0.0097	0.0163
Massachusetts	ND	-	-	-	-	-	-	-
Maryland	0.0020	-	0.0015	0.0001	0.0006	-	-	0.0042
Maine	ND	-	-	0.0001	-	-	-	0.0001
Michigan	0.0071	-	0.0049	0.0000	0.0018	-	-	0.0138
Minnesota	0.0286	-	0.0204	0.0003	0.0040	-	-	0.0533
Missouri	0.0123	-	0.0143	-	0.0021	0.0015	-	0.0302
Mississippi	0.0018	-	0.0028	-	0.0006	0.0165	-	0.0217
Montana	0.0001	-	-	0.0011	0.0048	-	-	0.0060
North Carolina	0.0028	0.0002	0.0033	0.0000	0.0009	-	-	0.0073
North Dakota	0.0029	-	0.0055	0.0030	0.0146	-	-	0.0260
Nebraska	0.0404	-	0.0171	0.0000	0.0030	-	-	0.0605
New Hampshire	ND	-	-	-	-	-	-	-
New Jersey	0.0003	-	0.0002	0.0000	0.0001	-	-	0.0006
New Mexico	0.0003	0.0000	-	-	0.0004	-	-	0.0007
Nevada	ND	-	-	0.0000	0.0000	-	-	0.0000
New York	0.0020	-	0.0004	0.0000	0.0003	-	-	0.0028
Ohio	0.0155	-	0.0144	0.0000	0.0030	-	-	0.0330
Oklahoma	0.0009	0.0001	0.0004		0.0061	-	-	0.0075
Oregon	0.0001	-	-	0.0002	0.0017	-	-	0.0019
Pennsylvania	0.0034	-	0.0011	0.0002	0.0004	-	-	0.0051
Rhode Island	ND	-	-	-	-	-	-	-
South Carolina	0.0009	0.0000	0.0007		0.0005	-	-	0.0021
South Dakota	0.0132	-	0.0106	0.0002	0.0038	-	-	0.0278
Tennessee	0.0029	-	0.0027	-	0.0009	-	-	0.0066
Texas	0.0059	0.0005	0.0004		0.0054	-	0.0010	0.0133
Utah	0.0001	-	-	0.0002	0.0003	-	-	0.0005
Virginia	0.0014	0.0001	0.0013	0.0001	0.0005	-	-	0.0036
Vermont	ND	-	-	-	-	-	-	-
Washington	0.0004	-	-	0.0008	0.0066	-	-	0.0078
Wisconsin	0.0117	-	0.0046	0.0001	0.0005	-	-	0.0169
West Virginia	0.0001	-	0.0001	-	0.0000	-	-	0.0002
Wyoming	0.0002	-	-	0.0003	0.0002	-	-	0.0006
United States	0.3374	0.0026	0.2218	0.0093	0.0978	0.0686	0.0227	0.7601

[1] ND = no data.

[2] - = zero.

Appendix Table B-5 State nitrous oxide emissions from agriculture burning by crop type for 2001

	Corn	Peanuts	Soybeans	Barley	Wheat	Rice	Sugarcane	Total
				Tg CO_2 eq.				
Alaska	ND[1]	ND	ND	ND	ND	ND	ND	ND
Alabama	0.0002	0.0002	0.0005	-[2]	0.0001	-	-	0.0009
Arkansas	0.0003	-	0.0087	-	0.0009	0.0118	-	0.0216
Arizona	0.0001	-	-	0.0001	0.0001	-	-	0.0003
California	0.0003	-	-	0.0001	0.0006	0.0102	-	0.0111
Colorado	0.0017	-	-	0.0001	0.0012	-	-	0.0030
Connecticut	ND	-	-	-	-	-	-	-
Delaware	0.0003	-	0.0007	0.0000	0.0001	-	-	0.0011
Florida	0.0000	0.0001	0.0000	-	0.0000	-	0.0025	0.0026
Georgia	0.0003	0.0006	0.0004	-	0.0002	-	-	0.0015
Hawaii	-	-	-	-	-	-	0.0003	0.0003
Iowa	0.0187	-	0.0460	-	0.0000	-	-	0.0646
Idaho	0.0001	-	-	0.0008	0.0015	-	-	0.0023
Illinois	0.0185	-	0.0457	-	0.0007	-	-	0.0650
Indiana	0.0099	-	0.0262	-	0.0004	-	-	0.0365
Kansas	0.0043	-	0.0084	0.0000	0.0056	-	-	0.0183
Kentucky	0.0018	-	0.0047	0.0000	0.0004	-	-	0.0068
Louisiana	0.0005	-	0.0019	-	0.0001	0.0014	0.0022	0.0062
Massachusetts	ND	-	-	-	-	-	-	-
Maryland	0.0006	-	0.0019	0.0001	0.0002	-	-	0.0028
Maine	ND	-	-	0.0000	-	-	-	0.0000
Michigan	0.0022	-	0.0061	0.0000	0.0006	-	-	0.0090
Minnesota	0.0090	-	0.0255	0.0001	0.0014	-	-	0.0360
Missouri	0.0039	-	0.0178	-	0.0007	0.0007	-	0.0231
Mississippi	0.0006	-	0.0035	-	0.0002	0.0076	-	0.0119
Montana	0.0000	-	-	0.0005	0.0016	-	-	0.0021
North Carolina	0.0009	0.0001	0.0041	0.0000	0.0003	-	-	0.0055
North Dakota	0.0009	-	0.0069	0.0012	0.0050	-	-	0.0140
Nebraska	0.0128	-	0.0213	0.0000	0.0010	-	-	0.0351
New Hampshire	ND	-	-	-	-	-	-	-
New Jersey	0.0001	-	0.0003	0.0000	0.0000	-	-	0.0004
New Mexico	0.0001	0.0000	-	-	0.0001	-	-	0.0003
Nevada	ND	-	-	0.0000	0.0000	-	-	0.0000
New York	0.0006	-	0.0005	0.0000	0.0001	-	-	0.0013
Ohio	0.0049	-	0.0180	0.0000	0.0010	-	-	0.0239
Oklahoma	0.0003	0.0001	0.0005	ND	0.0021	-	-	0.0029
Oregon	0.0000	-	-	0.0001	0.0006	-	-	0.0007
Pennsylvania	0.0011	-	0.0014	0.0001	0.0001	-	-	0.0027
Rhode Island	ND	-	-	-	-	-	-	-
South Carolina	0.0003	0.0000	0.0009	ND	0.0002	-	-	0.0014
South Dakota	0.0042	-	0.0133	0.0001	0.0013	-	-	0.0188
Tennessee	0.0009	-	0.0034	-	0.0003	-	-	0.0046
Texas	0.0019	0.0003	0.0005	ND	0.0019	-	0.0002	0.0048
Utah	0.0000	-	-	0.0001	0.0001	-	-	0.0002
Virginia	0.0005	0.0001	0.0017	0.0001	0.0002	-	-	0.0024
Vermont	ND	-	-	-	-	-	-	-
Washington	0.0001	-	-	0.0003	0.0023	-	-	0.0027
Wisconsin	0.0037	-	0.0057	0.0000	0.0002	-	-	0.0096
West Virginia	0.0000	-	0.0001	-	0.0000	-	-	0.0001
Wyoming	0.0001	-	-	0.0001	0.0001	-	-	0.0002
United States	0.1066	0.0015	0.2765	0.0039	0.0334	0.0316	0.0052	0.4588

[1] ND = no data.
[2] - = zero.

Appendix Table B-6 Area harvested for rice cultivation, 1990-2001

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
						hectares						
Arkansas	485,633	509,915	558,478	497,774	574,666	542,291	473,493	562,525	601,174	657,831	570,619	656,010
Primary	485,633	509,915	558,478	497,774	574,666	542,291	473,493	562,525	600,971	657,628	570,619	656,010
Ratoon	0	0	0	0	0	0	0	0	202	202	0	0
California	159,854	144,071	159,450	176,851	196,277	188,183	202,347	208,822	185,350	204,371	221,773	190,611
Florida	7,467	12,869	13,962	13,962	14,569	14,569	13,355	11,534	12,141	11,902	10,994	7,314
Primary	4,978	8,580	9,308	9,308	9,713	9,713	8,903	7,689	8,094	7,229	7,801	4,562
Ratoon	2,489	4,290	4,654	4,654	4,856	4,856	4,452	3,845	4,047	4,673	3,193	2,752
Louisiana	286,726	268,312	326,184	278,834	326,184	299,879	280,413	306,718	326,184	324,079	271,955	287,252
Primary	220,558	206,394	250,911	214,488	250,911	230,676	215,702	235,937	250,911	249,292	194,253	220,963
Ratoon	66,168	61,918	75,273	64,346	75,273	69,203	64,711	70,781	75,273	74,788	77,701	66,289
Mississippi	101,174	89,033	111,291	99,150	126,669	116,552	84,176	96,317	108,458	130,716	88,223	102,388
Missouri	32,376	37,232	45,326	37,637	50,182	45,326	38,446	47,349	57,871	74,464	68,393	83,772
Texas	200,000	194,334	198,867	168,839	200,567	180,170	168,839	146,742	160,340	146,742	129,907	122,380
Primary	142,857	138,810	142,048	120,599	143,262	128,693	120,599	104,816	114,529	104,816	86,605	87,414
Ratoon	57,143	55,524	56,819	48,240	57,305	51,477	48,240	41,926	45,811	41,926	43,302	34,966
Total	1,273,229	1,255,767	1,413,557	1,273,047	1,489,114	1,386,969	1,261,068	1,380,008	1,451,518	1,550,106	1,361,864	1,449,726

Source: USDA NASS Field Crops (1994, 1998); USDA NASS Crop Production (2001, 2002).

Appendix Table B-7 Methane from rice cultivation for primary and ratoon operations, by State, 1990-2001

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
						Tg CO_2 eq.						
Primary	5.06	5.00	5.63	5.10	5.96	5.56	5.04	5.57	5.85	6.30	5.46	5.93
Arkansas	2.14	2.25	2.46	2.20	2.53	2.39	2.09	2.48	2.65	2.90	2.52	2.89
California	0.70	0.64	0.70	0.78	0.87	0.83	0.89	0.92	0.82	0.90	0.98	0.84
Florida	0.02	0.04	0.04	0.04	0.04	0.04	0.04	0.03	0.04	0.03	0.03	0.02
Louisiana	0.97	0.91	1.11	0.95	1.11	1.02	0.95	1.04	1.11	1.10	0.86	0.97
Mississippi	0.45	0.39	0.49	0.44	0.56	0.51	0.37	0.42	0.48	0.58	0.39	0.45
Missouri	0.14	0.16	0.20	0.17	0.22	0.20	0.17	0.21	0.26	0.33	0.30	0.37
Texas	0.63	0.61	0.63	0.53	0.63	0.57	0.53	0.46	0.51	0.46	0.38	0.39
Ratoon	2.06	1.99	2.24	1.92	2.25	2.06	1.92	1.91	2.05	1.99	2.03	1.70
Arkansas	+[1]	+	+	+	+	+	+	+	+	+	+	+
Florida	0.04	0.07	0.08	0.08	0.08	0.08	0.07	0.06	0.07	0.08	0.05	0.05
Louisiana	1.08	1.01	1.23	1.05	1.23	1.13	1.06	1.16	1.23	1.23	1.27	1.09
Texas	0.94	0.91	0.93	0.79	0.94	0.84	0.79	0.69	0.75	0.69	0.71	0.57

[1] + < 0.005.

Appendix Table B-8 Estimated ratios for nitrogen fixing crops and crop residue

Crop	Residue/ crop ratio	Residue dry matter fraction	Residue nitrogen fraction
Soybeans	2.1	0.87	0.023
Peanuts	1	0.86	0.0106
Dry edible beans	1.55	0.87	0.0062
Dry edible peas	1.55	0.87	0.0062
Austrian winter peas	1.55	0.87	0.0062
Lentils	1.55	0.87	0.0062
Wrinkled seed peas	1.55	0.87	0.0062
Alfalfa	0	0.85	NA
Corn	1	0.91	0.0058
Wheat	1.3	0.93	0.0062
Barley	1.2	0.93	0.0077
Sorghum	1.4	0.91	0.0108
Oats	1.3	0.92	0.007
Rye	1.6	0.9	0.0048
Millet	1.4	0.89	0.007
Rice	1.4	0.91	0.0072

Source: EPA (2003a).

Appendix Table B-9 Aboveground biomass nitrogen in nitrogen-fixing crops, 1990-2001

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
						Gg N						
Soybeans	4,240.95	4,374.38	4,823.18	4,117.14	5,537.77	4,787.73	5,241.39	5,920.65	6,035.74	5,843.60	6,072.72	6,365.07
White clover	2,734.67	2,734.67	2,734.67	2,734.67	2,734.67	2,734.67	2,734.67	2,734.67	2,734.67	2,734.67	2,734.67	2,734.67
Alfalfa	1,730.44	1,728.66	1,642.00	1,662.28	1,683.07	1,745.70	1,641.88	1,655.39	1,707.70	1,739.90	1,641.90	1,646.21
Red clover	513.00	513.00	513.00	513.00	513.00	513.00	513.00	513.00	513.00	513.00	513.00	513.00
Birdsfoot trefoil	99.00	99.00	99.00	99.00	99.00	99.00	99.00	99.00	99.00	99.00	99.00	99.00
Dry edible beans	97.75	101.93	68.27	66.00	87.40	92.65	84.26	88.66	91.83	99.88	79.73	58.99
Peanuts	84.34	115.31	100.28	79.40	99.41	81.02	85.69	82.84	92.77	89.63	76.43	99.23
Arrowleaf clover	66.78	66.78	66.78	64.70	62.61	60.52	58.44	56.35	54.26	52.18	48.00	50.09
Crimson clover	20.54	20.54	20.54	19.29	18.04	16.79	15.54	14.29	13.03	11.78	10.53	9.28
Dry edible peas	7.16	11.22	7.65	9.94	6.81	14.39	8.06	17.36	17.91	14.41	10.56	11.41
Wrinkled seed peas	2.78	2.79	1.62	2.56	2.28	3.16	1.65	2.06	2.03	1.99	2.05	1.93
Lentils	2.64	5.04	4.73	6.06	5.60	6.71	4.02	7.26	5.85	7.21	9.14	8.75
Austrian winter peas	0.38	0.42	0.30	0.47	0.15	0.36	0.31	0.35	0.31	0.18	0.22	0.29

Source: EPA (2003a).

Appendix Table B-10 Nitrogen in crop residues applied to soils

	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001
						Gg N						
Soybeans	1,982.30	2,044.67	2,254.45	1,924.43	2,588.46	2,237.88	2,449.92	2,767.43	2,821.22	2,731.41	2,838.51	2,975.16
Corn	957.32	901.91	1,143.46	764.71	1,212.70	892.89	1,114.01	1,110.90	1,177.49	1,137.90	1,196.36	1,147.10
All wheat	501.19	363.56	452.91	439.99	426.14	400.75	418.13	455.60	467.69	422.10	409.88	359.43
Sorghum	180.33	183.97	275.24	168.02	203.12	144.27	250.15	199.28	163.54	187.21	148.00	161.84
Barley	71.09	78.19	76.63	67.02	63.12	60.51	66.08	60.60	59.29	47.20	53.67	42.03
Rice	51.49	53.36	60.89	52.30	65.36	59.68	57.44	65.78	69.18	74.87	68.60	88.85
Oats	39.12	26.67	32.18	22.61	25.03	17.62	16.76	18.29	18.15	15.99	16.36	12.78
Peanuts	13.41	18.33	15.94	12.62	15.81	12.88	13.62	13.17	14.75	14.25	12.15	15.78
Dry edible beans	11.05	11.52	7.72	7.46	9.88	10.47	9.53	10.02	10.38	11.29	9.01	6.67
Millet	3.19	3.19	3.19	3.19	3.19	3.19	3.19	3.19	3.19	3.19	1.30	3.43
Rye	1.61	1.54	1.81	1.63	1.79	1.59	1.41	1.28	1.92	1.74	1.33	1.10
Dry edible peas	0.81	1.27	0.87	1.12	0.77	1.63	0.91	1.96	2.03	1.63	1.19	1.29
Wrinkled seed peas	0.31	0.32	0.18	0.29	0.26	0.36	0.19	0.23	0.23	0.22	0.23	0.22
Lentils	0.30	0.57	0.53	0.68	0.63	0.76	0.45	0.82	0.66	0.81	1.03	0.99
Austrian winter peas	0.04	0.05	0.03	0.05	0.02	0.04	0.04	0.04	0.04	0.02	0.02	0.03

Source: EPA (2003a).

Appendix Table B-11 State estimates of soil carbon changes in cropland and grazing land in 1997 by major activity categories

State	Plowout of grassland to annual cropland[1]	Cropland manage-ment	Other cropland[2]	Cropland converted to hayland[3]	Hayland manage-ment	Cropland converted to grazing land[3]	Grazing land man-age-ment	CRP	Manure applica-tion	Cultiva-tion of organic soils	Net soil carbon emis-sions[4]
						Tg CO_2 eq.					
Alabama	0.37	(0.11)	(0.04)	(0.26)	0.00	(0.62)	0.00	(0.29)	(0.53)	0.04	(1.45)
Alaska	n.d.[5]	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
Arizona	0.22	0.00	(0.04)	(0.15)	0.00	(0.04)	0.00	(0.04)	(0.19)	0.00	(0.23)
Arkansas	0.81	(0.22)	(0.81)	(0.11)	0.04	(0.37)	(0.11)	(0.15)	(0.88)	0.00	(1.80)
California	1.14	(0.04)	(0.48)	(0.55)	0.00	(0.48)	0.04	(0.11)	(1.72)	2.35	0.15
Colorado	0.77	(0.15)	0.00	(0.55)	(0.04)	(0.26)	0.00	(1.25)	(0.53)	0.00	(2.00)
Connecticut	0.04	0.00	0.00	(0.07)	0.00	0.00	0.00	0.00	(0.04)	0.04	(0.04)
Delaware	0.04	(0.04)	0.00	0.00	0.00	0.00	0.00	0.00	(0.15)	0.00	(0.15)
Florida	0.33	(0.04)	(0.22)	(0.07)	0.00	(0.48)	0.11	(0.07)	(0.23)	10.74	10.07
Georgia	0.29	(0.07)	(0.04)	(0.26)	0.00	(0.48)	0.00	(0.18)	(0.76)	0.07	(1.42)
Hawaii	0.00	0.00	(0.04)	0.00	0.00	(0.04)	0.00	0.00	n.d.	0.29	0.22
Idaho	1.10	(0.07)	0.00	(1.03)	(0.04)	(0.26)	(0.04)	(0.59)	(0.34)	0.07	(1.19)
Illinois	3.08	(0.48)	(0.04)	(1.58)	0.00	(0.51)	0.04	(0.59)	(0.53)	0.84	0.24
Indiana	1.61	(0.55)	(0.04)	(0.88)	(0.04)	(0.44)	0.00	(0.26)	(0.61)	1.98	0.78
Iowa	4.44	(0.11)	0.00	(1.91)	0.00	(0.59)	0.04	(0.77)	(1.49)	1.87	1.48
Kansas	2.05	(0.99)	0.00	(1.32)	0.00	(0.70)	0.00	(1.54)	(0.88)	0.00	(3.37)
Kentucky	0.95	(0.11)	(0.07)	(0.84)	(0.04)	(0.77)	0.00	(0.11)	(0.15)	0.00	(1.14)
Louisiana	1.14	(0.22)	(1.03)	(0.07)	0.00	(0.37)	(0.07)	(0.11)	(0.04)	0.07	(0.70)
Maine	0.11	0.00	0.00	(0.11)	0.00	(0.07)	(0.04)	0.00	(0.08)	0.00	(0.19)
Maryland	0.18	(0.04)	0.00	(0.15)	0.00	(0.07)	0.00	0.00	(0.08)	0.00	(0.15)
Massachusetts	0.07	0.00	0.00	(0.07)	0.00	0.00	(0.04)	0.00	(0.19)	0.07	(0.15)
Michigan	2.09	(0.07)	(0.07)	(1.72)	(0.07)	(0.51)	0.00	(0.15)	(0.46)	3.12	2.14
Minnesota	4.62	0.00	(0.04)	(3.01)	(0.11)	(0.55)	0.04	(0.95)	(1.18)	5.24	4.06
Mississippi	0.88	(0.22)	(0.66)	(0.22)	0.00	(0.62)	0.00	(0.40)	(0.42)	0.04	(1.63)
Missouri	1.91	(0.44)	(0.26)	(1.14)	0.11	(0.84)	0.15	(0.70)	(0.65)	0.04	(1.82)
Montana	1.91	(0.59)	0.00	(1.28)	(0.07)	(0.48)	0.00	(1.80)	(0.08)	0.11	(2.28)
Nebraska	3.08	(0.66)	0.00	(2.09)	(0.04)	(0.33)	0.04	(0.99)	(1.03)	0.00	(2.02)
Nevada	0.11	0.00	0.00	(0.18)	(0.04)	(0.07)	0.04	0.00	(0.04)	0.00	(0.18)
New Hampshire	0.04	0.00	0.00	(0.07)	0.07	0.00	0.00	0.00	0.00	0.04	0.07
New Jersey	0.11	(0.04)	0.00	(0.11)	0.00	(0.04)	0.00	0.00	(0.04)	0.04	(0.07)
New Mexico	0.26	0.00	0.00	(0.22)	0.00	(0.18)	0.00	(0.22)	(0.23)	0.00	(0.60)
New York	1.61	(0.04)	(0.04)	(2.27)	(0.04)	(0.26)	(0.04)	0.00	(0.61)	0.48	(1.20)
North Carolina	0.22	(0.07)	(0.04)	(0.22)	0.00	(0.33)	0.00	(0.11)	(1.26)	1.06	(0.75)
North Dakota	3.15	(1.28)	0.00	(2.27)	0.00	(0.33)	0.04	(2.71)	(0.11)	0.22	(3.30)
Ohio	1.87	(0.40)	(0.04)	(1.47)	(0.04)	(0.40)	0.00	(0.22)	(0.53)	1.25	0.02
Oklahoma	0.88	(0.44)	0.00	(0.40)	0.00	(0.99)	(0.07)	(0.77)	(0.46)	0.00	(2.26)
Oregon	0.33	(0.04)	(0.04)	(0.44)	0.00	(0.18)	(0.07)	(0.29)	(0.15)	0.18	(0.70)
Pennsylvania	1.21	(0.07)	(0.04)	(1.43)	0.00	(0.18)	(0.04)	0.00	(0.80)	0.07	(1.28)
Rhode Island	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
South Carolina	0.15	(0.04)	0.00	(0.15)	0.04	(0.18)	0.00	(0.07)	(0.19)	0.62	0.18
South Dakota	4.07	(0.18)	0.00	(2.90)	(0.04)	(0.44)	0.07	(1.39)	(0.31)	0.07	(1.04)
Tennessee	0.70	(0.07)	(0.18)	(0.59)	(0.07)	(0.77)	(0.11)	(0.11)	(0.15)	0.00	(1.36)
Texas	3.01	(0.59)	(0.37)	(0.33)	0.00	(3.41)	0.00	(2.13)	(1.53)	0.00	(5.34)
Utah	0.29	0.00	0.00	(0.55)	(0.04)	(0.15)	0.00	(0.15)	(0.15)	0.00	(0.74)
Vermont	0.07	0.00	0.00	(0.11)	0.00	0.00	(0.04)	0.00	(0.11)	0.00	(0.19)
Virginia	0.37	(0.07)	0.00	(0.40)	0.00	(0.33)	(0.04)	(0.07)	(0.34)	0.40	(0.49)
Washington	0.51	(0.15)	(0.11)	(0.51)	(0.04)	(0.18)	(0.07)	(0.81)	(0.27)	0.22	(1.40)
West Virginia	0.26	0.00	0.00	(0.37)	0.00	(0.11)	(0.04)	0.00	(0.08)	0.00	(0.33)
Wisconsin	4.84	0.00	0.00	(3.92)	(0.07)	(0.40)	0.04	(0.15)	(1.30)	2.93	1.96
Wyoming	0.51	(0.07)	0.00	(0.62)	(0.04)	(0.29)	0.00	(0.37)	(0.04)	0.00	(0.92)
Totals	57.82	(8.84)	(4.69)	(38.98)	(0.59)	(19.10)	(0.18)	(20.61)	(21.97)	34.58	(22.52)

See next page for footnotes

Footnotes for Appendix Table B-11

Parentheses indicate net sequestration.

[1] Losses from annual cropping systems due to plow-out of pastures, rangeland, hayland, set-aside lands, and perennial/horticultural cropland (annual cropping systems on mineral soils, e.g., corn, soybean, cotton, and wheat).

[2] Perennial/horticultural cropland and rice cultivation.

[3] Gains in soil carbon sequestration due to land conversions from annual cropland into hay or grazing land.

[4] Total does not include change in soil organic carbon storage on federal lands, including those that were previously under private ownership, and does not include carbon storage due to sewage sludge applications.

[5] No data.

Appendix Table B-12 U.S. soil groupings, based on the IPCC categories and dominant taxonomic soil, and reference carbon stocks

IPCC Inventory soil categories	USDA Taxonomic soil orders	Climate regions CTD	CTM	WTD	WTM	STD	STM
		Tg C/ha					
High clay activity mineral soils	Vertisols, mollisols, inceptisols, aridisols, and high base status alfisols	42 (n=133)	65 (n=526)	37 (n=203)	51 (n=424)	42 (n=26)	57 (n=12)
Low clay activity mineral soils	Ultisols, oxisols, acidic alfisols, and many entisols	45 (n=37)	52 (n=113)	25 (n=86)	40 (n=300)	39 (n=13)	47 (n=7)
Sandy soils	Any soils with greater than 70% sand and less than 8% clay (often Entisols)	24 (n=5)	40 (n=43)	16 (n=19)	30 (n=102)	33 (n=186)	50 (n=18)
Volcanic soils	Andisols	124 (n=12)	114 (n=2)	124 (n=12)	124 (n=12)	124 (n=12)	128 (n=9)
Spodic soils	Spodosols	86 (n=20)	74 (n=13)	86 (n=20)	107 (n=7)	86 (n=20)	86 (n=20)
Aquic soils	Soils with aquic suborder	86 (n=4)	89 (n=161)	48 (n=26)	51 (n=300)	63 (n=503)	48 (n=12)
Organic soils	Histosols	NA[1]	NA[1]	NA[1]	NA[1]	NA[1]	NA[1]

Source: Carbon stocks estimated from pedon data in the NSSC database [USDA NRCS (1997)] for top 30 cm of the soil profile; sample size provided in parentheses.

[1] Carbon stocks are not needed for organic soils.

Appendix Table B-13 Management factors for the U.S. and the IPCC default values

	IPCC default	U.S. factor
Land use change		
Cultivated[1]	1	1
General uncultivated[1,2]	1.4	1.3 (±0.04)
Set-aside[1]	1.25	1.2 (±0.03)
Improved pasture lands[3]	1.1	1.1
Wetland rice production[3]	1.1	1.1
Tillage		
Conventional till	1	1
Reduced till	1.05	1.02 (±0.03)
No-till	1.1	1.13 (±0.03)
Input		
Low	0.9	0.94 (±0.01)
Medium	1	1
High	1.1	1.07 (±0.03)

[1] Factors in the IPCC (IPCC/UNEP/OECD/IEA 1997) documentation were converted to represent changes in SOC storage from a cultivated condition rather than a native condition.
[2] Default factor was higher for aquic soils at 1.7, but the U.S. analysis showed no significant differences between aquic and non-aquic soils and so a single U.S. factor was estimated for all soil types.
[3] A U.S. specific factor was not estimated for land or management leading to additional carbon storage because of few studies addressing the impact of legume mixtures, irrigation, or manure applications for pasture lands in the United States, or the impact of wetland rice production in the United States.

Appendix Table B-14 Land use and management categories

General land use category	Specific management sub-categories	IPCC Category Mineral soils	 Organic soils
Agricultural (cropland and grazing land)	Irrigated crops	High input cultivation	Cultivated crops
	Continuous row crops	Medium input cultivation	Cultivated crops
	Continuous small grains	Medium input cultivation	Cultivated crops
	Continuous row crops and small grains	Medium input cultivation	Cultivated crops
	Row crops in rotation with hay and/or pasture	High input cultivation	Cultivated crops
	Small grains in rotation with hay and/or pasture	High input cultivation	Cultivated crops
	Row crops and small grains in rotation with hay and/or pasture	High input cultivation	Cultivated crops
	Vegetable crops	Low input cultivation	Cultivated crops
	Low residue annual crops (e.g., tobacco or cotton)	Low input cultivation	Cultivated crops
	Small grains with fallow	Low input cultivation	Cultivated crops
	Row crops and small grains with fallow	Low input cultivation	Cultivated crops
	Row crops with fallow	Low input cultivation	Cultivated crops
	Miscellaneous crop rotations	Medium input cultivation	Cultivated crops
	Continuous rice	Improved land[1]	Undrained
	Rice in rotation with other crops	Improved land[1]	Undrained
	Continuous perennial or horticultural crops	Improved land[1]	Pasture/forest
	Continuous hay	Uncultivated land (general)	Pasture/forest
	Continuous hay with legumes or irrigation	Improved land[1]	Pasture/forest
	CRP	Uncultivated land (set-aside)	Undrained
	Rangeland	Uncultivated land (general)	Undrained
	Continuous pasture	Uncultivated land (general)	Pasture/forest
	Continuous pasture with legumes or irrigation	Improved land[1]	Pasture/forest
	Aquaculture[2]	Not estimated	Not estimated
Non-agricultural[3]	Forest	Uncultivated land (general)	Pasture/forest
	Federal	Uncultivated land (general)	Undrained
	Water[2]	Not estimated	Not estimated
	Urban land[2]	Not estimated	Not estimated
	Miscellaneous[2,4]	Not estimated	Not estimated

Source: (USDA NRCS 2000).

[1] Improved land increases SOC storage above the levels found in general land-use changes.

[2] Assume no change in carbon stocks when converting to or from these land uses because of a lack of information about the effect of these practices on SOC storage.

[3] Some non-agricultural land is included in the inventory because it was an agricultural land use in 1992 or 1997.

[4] Includes a variety of land uses from roads, beaches, and marshes to mining and gravel pits.

Appendix Table B-15 Tillage percentages for each management category in the U.S. by climate zones, with adjustments for long-term adoption of no-till agriculture

	1982			1992			1997		
System	No till[1]	Red. till[2]	Conv. till[3]	No till[1]	Red. till[2]	Conv. till[3]	No till[1]	Red. till[2]	Conv. till[3]
STD									
Continuous cropping rotations†	0	3	97	0	4	96	0	15	85
Rotations with fallow ‡	0	0	100	0	2	98	0	5	95
Low residue ag. *	0	3	97	0	4	96	0	10	90
STM									
Continuous cropping rotations†	0	0	100	0	20	80	1	10	89
Rotations with fallow ‡	0	0	100	0	10	90	1	10	89
Low residue ag. *	0	3	97	0	4	96	0	5	95
WTD									
Continuous cropping rotations†	0	0	100	0	10	90	1	15	84
Rotations with fallow ‡	0	3	97	0	15	85	2	20	78
Low residue ag. *	0	3	97	0	1	99	0	0	100
WTM									
Continuous cropping rotations†	0	6	94	10	30	60	12	28	60
Rotations with fallow ‡	0	6	94	5	30	65	8	27	65
Low residue ag. *	0	9	91	1	10	89	2	13	85
CTD									
Continuous cropping rotations†	0	3	97	2	25	73	8	12	80
Rotations with fallow ‡	0	6	94	4	25	71	12	13	75
Low residue ag. *	0	0	100	1	2	97	2	6	92
CTM									
Continuous cropping rotations†	0	11	89	5	30	65	3	17	80
Rotations with fallow ‡	0	11	89	5	30	65	3	27	70
Low residue ag. *	0	0	100	1	2	97	1	7	92

[1] No-till includes CTIC survey data designated as no-tillage.
[2] Reduced-till includes CTIC survey data designated as ridge tillage, mulch tillage, and reduced tillage.
[3] Conventional till includes CTIC survey data designated as intensive tillage and conventional tillage.
† Medium and high input rotations (based on the IPCC categories) found in Table B-14. CTIC survey data for corn, soybeans, and sorghum were used in this category.
‡ Rotations with fallow found in Table B-14. CTIC survey data on fallow and small grain cropland were used in this category.
* Low input rotations found in Table B-14, with the exception of rotations with fallow. CTIC survey data on cotton were used in this category; tillage rates are assumed to be the same for low residue crops and vegetables in rotation.

Appendix Table B-16 Carbon loss rates from organic soils under agricultural management in the U.S. and the IPCC default rates

Region	Cropland	Pasture/Forest[1]
	Tg C/ha-yr	
CTD & CTM	11.2±2.5	2.8±0.51
WTD & WTM	14.0±2.5	3.5±0.81
STD & STM	14.0±3.3	3.5±0.81

[1] There were not enough data available to estimate a U.S. value for C losses from managed pastures and forests. Consequently, estimates are 25% of the values for cropland, which was an assumption used for the IPCC default organic soil C losses on pasture/ forest lands.

Appendix C: Forest Carbon Stocks

Appendix Table C-1 State summaries of forest area, carbon stocks for 2002, estimated net forest stock change 2001, and estimated net harvested wood products stock change for 2001. Total stock change per State is the sum of the forest and products estimates.

State	Area	Forest non-soil stocks	Forest non-soil stock change	Products stock change
	1,000 ha	*Tg CO_2 eq.*	*Tg CO_2 eq./yr*	*Tg CO_2 eq./yr*
Alabama	9,302	2,540	(24.4)	(19.3)
Alaska	nd[1]	nd	nd	nd
Arizona	7,862	1,801	(2.4)	(0.2)
Arkansas	7,596	2,191	(23.4)	(10.0)
California	16,282	7,527	(21.4)	(8.4)
Colorado	8,756	3,214	(3.5)	(0.3)
Connecticut	752	324	(1.3)	(0.2)
Delaware	155	61	(0.4)	(0.1)
Florida	6,590	1,429	(11.6)	(7.8)
Georgia	9,876	2,625	(26.9)	(20.2)
Hawaii	nd	nd	nd	nd
Idaho	8,760	4,145	(12.1)	(3.4)
Illinois	1,753	674	(2.6)	(1.0)
Indiana	1,821	744	(11.6)	(1.3)
Iowa	830	244	(3.8)	(0.3)
Kansas	625	179	(2.4)	(0.1)
Kentucky	5,133	1,738	(10.8)	(3.3)
Louisiana	5,589	1,707	(8.2)	(10.6)
Maine	7,163	2,582	(3.2)	(5.9)
Maryland	1,038	454	(2.3)	(0.5)
Massachusetts	1,265	563	(3.4)	(0.2)
Michigan	7,803	2,998	(23.6)	(4.7)
Minnesota	6,750	2,016	(8.6)	(4.3)
Mississippi	7,519	2,048	(22.3)	(15.5)
Missouri	5,662	1,390	(12.2)	(2.2)
Montana	9,426	3,938	(21.5)	(2.3)
Nebraska	383	111	(1.8)	(0.1)
Nevada	4,130	856	(9.6)	(0.0)
New Hampshire	1,950	887	(7.1)	(1.9)
New Jersey	863	304	(3.4)	(0.1)
New Mexico	6,751	1,802	1.7	(0.2)
New York	7,459	2,548	(15.5)	(1.9)
North Carolina	7,811	2,558	(15.6)	(13.7)
North Dakota	272	57	(1.3)	(0.0)
Ohio	3,179	1,142	(17.6)	(1.4)
Oklahoma	3,102	625	(9.2)	(1.7)
Oregon	11,999	6,273	(29.4)	(11.5)
Pennsylvania	6,841	2,645	(8.2)	(2.9)
Rhode Island	156	56	(0.2)	(0.0)
South Carolina	5,056	1,390	(8.8)	(10.1)
South Dakota	655	192	0.6	(0.2)
Tennessee	5,826	2,046	(19.2)	(4.9)
Texas	6,940	1,591	(1.3)	(11.3)
Utah	6,344	1,736	1.4	(0.1)
Vermont	1,869	867	(11.7)	(1.0)
Virginia	6,505	2,195	(15.6)	(7.9)
Washington	8,818	5,263	(7.0)	(11.6)
West Virginia	4,900	1,947	(25.4)	(2.2)
Wisconsin	6,460	2,195	(15.0)	(4.8)
Wyoming	4,449	1,897	(7.8)	(0.2)

[1] no data; Parenthesis indicate net sequestration.

Appendix Table C-2 Carbon stocks on privately-owned forestland in 2001, by region, age-class, and pool

	Age class	Soil organic carbon	Non-living plant mass	Biomass	Total
			Tg CO_2 eq.		
North		26,060	5,997	13,155	45,212
	< 20	2,326	278	398	3,003
	20-40	4,136	626	1,229	5,990
	40-60	5,498	1,209	2,898	9,605
	60-80	5,502	1,453	3,383	10,338
	80-100	3,216	938	2,159	6,314
	100-150	1,802	578	1,174	3,553
	150-200	89	32	50	170
	200+	8	3	4	15
	Uneven	3,484	880	1,860	6,224
South		28,018	5,857	16,245	50,120
	< 20	9,302	1,279	2,768	13,348
	20-40	6,200	1,369	3,610	11,179
	40-60	6,213	1,622	4,800	12,635
	60-80	3,493	980	3,109	7,582
	80-100	942	281	904	2,127
	100-150	362	108	340	809
	150-200	9	2	8	20
	200+	0	0	0	0
	Uneven	1,498	217	705	2,420
Pacific Coast		4,749	3,085	4,001	11,835
	< 20	758	243	195	1,197
	20-40	663	408	554	1,624
	40-60	725	643	940	2,308
	60-80	561	479	693	1,733
	80-100	419	367	501	1,287
	100-150	280	275	352	906
	150-200	52	50	69	171
	200+	74	101	119	295
	Uneven	1,217	520	578	2,315
Rocky Mountain		3,924	1,926	2,181	8,032
	< 20	449	185	111	745
	20-40	218	83	75	376
	40-60	456	197	217	870
	60-80	835	418	511	1,764
	80-100	887	479	609	1,975
	100-150	623	338	403	1,364
	150-200	251	137	164	552
	200+	97	44	51	192
	Uneven	109	45	40	194

Note: Privately owned forests include timberlands, reserves, and other forestland.

Appendix Table C-3 Carbon stocks on publicly-owned forestland in 2001, by region, age-class, and pool

	Age class	Soil organic carbon	Non-living plant mass	Biomass	Total
			Tg CO_2 eq.		
North		9,486	1,831	3,817	15,133
	< 20	1,169	118	190	1,477
	20-40	1,244	156	293	1,693
	40-60	1,802	323	727	2,853
	60-80	1,775	409	953	3,137
	80-100	1,033	282	679	1,994
	100-150	540	160	323	1,022
	150-200	26	8	13	47
	200+	5	2	2	9
	Uneven	1,893	372	637	2,902
South		3,975	618	2,758	7,350
	< 20	617	63	199	879
	20-40	667	99	381	1,147
	40-60	962	169	743	1,874
	60-80	857	153	755	1,765
	80-100	276	52	257	585
	100-150	144	26	134	304
	150-200	9	2	8	19
	200+	1	0	2	3
	Uneven	442	54	278	774
Pacific Coast		7,487	3,905	7,867	19,259
	< 20	708	205	186	1,099
	20-40	337	124	233	695
	40-60	469	216	463	1,148
	60-80	659	325	654	1,639
	80-100	795	435	926	2,155
	100-150	1,573	907	1,828	4,307
	150-200	933	584	1,238	2,755
	200+	964	660	1,469	3,093
	Uneven	1,050	449	870	2,369
Rocky Mountain		12,805	6,136	8,939	27,880
	< 20	1,163	459	318	1,940
	20-40	490	182	159	831
	40-60	878	361	465	1,704
	60-80	1,629	746	1,154	3,529
	80-100	1,967	966	1,575	4,508
	100-150	2,948	1,495	2,299	6,741
	150-200	1,432	761	1,146	3,338
	200+	761	414	579	1,753
	Uneven	1,537	754	1,244	3,535

Note: Publicly owned forests include timberlands, reserves, and other forestland.

Appendix Table C-4 Carbon stocks on timberlands in 2001 by region, stand size class, and carbon pool

	Stand size class	Soil organic carbon	Non-living plant mass	Biomass	Total
			Tg CO_2 eq.		
North		33,135	7,392	16,250	56,777
	Nonstocked	70	7	6	83
	Seedling/ sapling	7,314	964	1,242	9,520
	Poletimber	10,865	2,418	5,004	18,287
	Sawtimber	14,887	4,003	9,998	28,887
South		30,229	6,288	18,158	54,674
	Nonstocked	255	20	40	315
	Seedling/ sapling	8,861	1,111	2,503	12,475
	Poletimber	7,598	1,626	4,462	13,686
	Sawtimber	13,514	3,530	11,152	28,197
Pacific Coast		8,154	5,168	8,817	22,139
	Nonstocked	240	57	46	343
	Seedling/ sapling	1,448	531	495	2,474
	Poletimber	902	422	641	1,965
	Sawtimber	5,564	4,159	7,635	17,357
Rocky Mountain		9,273	4,801	6,859	20,933
	Nonstocked	376	160	77	613
	Seedling/ sapling	1,049	435	275	1,759
	Poletimber	1,526	782	1,125	3,433
	Sawtimber	6,322	3,423	5,382	15,127

Note: Carbon stocks are for timberlands only.

Appendix Table C-5 Carbon stocks on all forestland for 2001 by forest type and ownership, excluding soils

	Private	Public	Reserve/ other
		Tg CO_2 eq.	
Eastern forests			
White-red-jack pine	1,207	392	87
Spruce-fir	1,673	660	221
Longleaf-slash pine	955	209	51
Loblolly-shortleaf pine	4,865	606	76
Oak-pine	3,200	462	138
Oak-hickory	14,717	2,210	855
Oak-gum-cypress	3,921	491	114
Elm-ash-cottonwood	1,541	310	82
Maple-beech-birch	7,361	1,506	356
Aspen-birch	1,145	645	98
Other eastern types	0	0	93
Nonstocked - East	29	6	21
Western forests			
Douglas-fir	2,836	5,365	1,075
Ponderosa pine	1,956	1,736	677
Western white pine	6	25	26
Fir-spruce	787	4,049	1,774
Hemlock-sitka spruce	777	1,226	648
Larch	70	176	32
Lodgepole pine	286	1,530	949
Redwood	342	34	49
Hardwoods	1,418	1,255	2,057
Other Western types	40	1,574	775
Pinyon-juniper	4	30	4,042
Chaparral	0	0	122
Nonstocked - West	35	63	166

Note: Private and public forestland include timberlands only.

Appendix Table C-6 Annual carbon stock change on all forestland for 2001 by forest type and ownership, excluding soils

	Private	Public	Reserve/Other
		Tg CO_2 eq.	
Eastern Forests			
White-red-jack pine	3.18	(0.61)	(7.26)
Spruce-fir	(2.40)	(2.58)	5.65
Longleaf-slash pine	(0.76)	(3.46)	(8.10)
Loblolly-shortleaf pine	(56.34)	(4.99)	(5.13)
Oak-pine	(3.76)	(0.94)	(15.66)
Oak-hickory	(102.73)	(29.27)	(58.82)
Oak-gum-cypress	(24.08)	(5.05)	(6.97)
Elm-ash-cottonwood	(17.59)	(3.70)	(0.63)
Maple-beech-birch	(30.06)	(12.58)	(1.75)
Aspen-birch	(7.29)	(3.12)	1.30
Other Eastern types	0.00	(0.02)	34.34
Nonstocked - East	8.92	0.70	26.28
Western forests			
Douglas-fir	4.33	58.58	30.98
Ponderosa pine	(2.74)	120.66	(23.13)
Western white pine	(0.04)	2.03	9.49
Fir-spruce	(4.29)	63.84	11.14
Hemlock-sitka spruce	(1.39)	(4.09)	5.32
Larch	0.04	(1.45)	(2.15)
Lodgepole pine	(1.11)	(17.00)	(35.62)
Redwood	(1.17)	0.29	(0.45)
Hardwoods	(3.63)	(49.37)	(126.86)
Other Western types	2.27	(243.04)	87.94
Pinyon-juniper	(0.68)	3.76	(42.57)
Chaparral	0.00	0.00	33.64
Nonstocked - West	(0.60)	(9.80)	(18.69)

Note: Private and public forestland include timberlands only.
Parenthesis indicate net sequestration.